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Ecological effects of large fires on US landscapes: benefit or catastrophe?

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Abstract: Many people assume that today's large fires are an ecological catastrophe because the perception is that they burn vast areas with high intensities and severities. However, little is known of the ecological impacts of large fires on historical and contemporary landscapes contributing to the high uncertainty and debate that surround both the causes and ecological consequences of these huge fires. This paper presents a review and summary of the current knowledge of the effect of large fires in United States ecosystems by important North American biomes that experienced large fires with each section written by a regional expert. The ecosystems covered are 1) ponderosa pine-Douglas-fir, 2) sagebrush-grasslands, 3) piñon-juniper, 4) chaparral, 5) mixed conifer, and 6) spruce-fir. Many important issues associated with large fires are addressed

including preconditioning factors, departure of current large fire effects with those that occurred historically, biotic responses to large fires, and comparisons across major biomes. This review has found that large fires were common on historical landscapes of the western US and they will continue to be common today. Sagebrush ecosystems are currently experiencing larger, more severe, and more frequent large fires compared to historical conditions due to exotic cheatgrass invasions. Large fires in southwest ponderosa pine forest historically created a mixed severity mosaic dominated by non-lethal surface fires while today's large fires are mostly high severity crown fires. This is quite different from the other ecosystems where historical and current large fire effects are similar. While large fires play an important role in landscape fire ecology for most regions, their importance is much less in the dry piñon juniper forests and sagebrush grasslands. Fire management must address the role of large fires in maintaining the health of many US fire-dominated ecosystems.

Keywords: Fire regimes, megafires, fire effects, large fires,

Introduction

Large wildland fires pose an interesting dilemma for fire management in North America. Many politicians, members of the public, and government agency land managers have come to believe that large wildfires (sometimes called megafires; fires > 10,000 ha) are an ecological disaster because are perceived to burn vast areas with high fire intensities and burn severities (Brown 1985, Daniel et al. 2007, Mutch et al. 1993, GAO 2002). On the other hand, these same megafires can return fire to deteriorating ecosystems where fires have been excluded for over 70 years thereby improving ecosystem health and reducing fire hazard (Agee 1998). Little is known of the ecological impacts, both short- and long-term, of large fires on historical and contemporary landscapes (Moreno 1998). This uncertainty fuels the debate that surrounds both the causes and ecological consequences of large fires which confuses the fire management community and the general public (Daniel et al. 2007). Just because a fire is large doesn't necessarily mean that it is unnatural or undesirable, unless, of course, it burns homes and threatens human life and property. Large fires may provide a unique opportunity for ecosystem restoration and fuel management treatments (van Wagtendonk 1995).

This paper presents a review and summary of the current knowledge of the effects of large fires in some major ecosystems of the United States. It is organized by those important biomes that typically experience large fires with each section written by a different author. Regions include the Pacific Northwest, Southern California, Northern Rockies, Southwestern United States, Great Basin, and Midwest with emphasis on the following ecosystems: ponderosa pine, Douglas-fir, sagebrush-grasslands, piñon-juniper, chaparral, Great Lakes conifer-hardwood forests, and spruce-fir. We attempt to address many important issues associated with large fires including 1) the scale of burn heterogeneity, 2) climate, vegetation, and topographical factors that may precondition large fire events, 3) departure of current large fire effects with those that occurred historically, 4) responses of key biota to large fires, and 5) broad generalities between and across major biomes. We also contrast effects of small fires and large fires in a spatial domain.

Background

Conventional wisdom holds that 70 years of fire exclusion, coupled with timber harvesting practices, and livestock grazing have tended to increase hazardous fuels that are now capable of fostering larger and more severe fires (Brown 1985, Arno and Brown 1991, Mutch et al. 1993, Kolb 1998, Keane et al. 2002, Pinol et al. 2005). In the absence of fire, subsequent vegetation development will generally result in an increase of canopy and surface fuels that will be more connected and continuously distributed across landscapes (Baker 1992, Ferry et al. 1995, Arno 1998). It follows that increased fuels may burn in intense, large fires that could kill most plants, propagules, and animals (Agee 1998) and alter many soil and biophysical site conditions (Ryan 2002). Regeneration of diverse post-fire plant communities on severely burned areas may be delayed or prevented because of adverse site conditions in the burned area, and this could increase the frequency and severity of erosional events (McNabb and Swanson 1990). However,

large fires were common on historical landscapes where the majority of burned area occurred during large fire events (Strauss et al. 1989, Malamud 2005, Cui and Perera 2006). In fact, the annual area burned during pre-Euro-American settlement large fire years is much larger than the annual area burned by today's large fires by an order of magnitude in some places (Arno 1980, Barrett et al. 1997). Some studies suggest that the area of unburned patches within large fire perimeter may actually increase with fire size (Eberhart and Woodward 1987), and that fire exclusion has had little effect on large fire dynamics and increased fuels don't seem to promote large fires (Johnson et al. 2001, Schoenberg et al. 2003, Bridge et al. 2005).

Many factors can precondition the regional landscape to experience large fires. Most large fires in the contiguous US occur in years of moderate to severe drought (Swetnam and Betancourt 1997, Heyerdahl et al. 2001, Baker 2003, Swetnam and Baisan 2003). This is especially true in topographically complex landscapes where, in normal climate years, subalpine to alpine ecosystems may stay moist throughout the year and thereby retard fire spread (Wadleigh and Jenkins 1996). Also, most area burned in large fires occurs during short-term wind events and/or under very hot and dry (low relative humidity) conditions (Cohen and Miller 1978, Schoennagel et al. 2007). Some evidence suggest that the amount and contagion of dead and live vegetation (fuels) on large landscapes can also contribute to large fires (Gardner et al. 1997, Turner et al. 1998), but this may not be true in all ecosystems and geographical regions due to the interactions of high winds, spotting, and large fire behaviour (Bessie and Johnson 1995, Schmoldt et al. 1999). Large fire ignition sources are mostly humans or lightning: lightning fires dominate in the western US while human-ignited fires are important in the southeastern US and California (Stephens 2005). Ignition dynamics (number, location, and source) are critically important in climate-fuel-fire interactions because without ignitions, large fires are impossible, even in extreme drought years (Ricotta et al. 1999).

The effects of large fires on ecosystems can be distinctly different from small fires. Large fires can create large burn patches (Agee 1998) that could slow wind and mammal dispersal of seed from unburned edges, thereby delaying common vegetation development processes. Large fires may be more severe because more fuel may be consumed and the additional heat generated can kill more above-ground vegetation and the deeper heat pulse will kill more below-ground biota (Ryan 2002). Large fires are more difficult and costly to fight, and the subsequent effects of those fires may be economically more damaging than small fires (Butry 2001, Calkin et al. 2005, Daniel et al. 2007). Last, large fires will tend to affect the most people and destroy the most property because of their shear size. However, severe large fires may be required on many landscapes to emulate historical fire dynamics and sustain healthy ecosystems (Fulé et al. 2004). Since large fires were common on most historical US landscapes, it follows that there are many plant and animal species that depend on the severity pattern created by large burns (Agee 1993, Debano et al. 1998, Habeck and Mutch 1973, Hutto 1995).

Regional Effects of Large Fires

Pacific Northwest

Large and severe fires are historically characteristic of Pacific Northwest forests. The section focuses on the forests west of the Cascade Mountains called the Douglas-fir region, which are influenced by a maritime climate that is wet and relatively warm compared to inland regions (Franklin and Dyrness 1973). Douglas-fir (*Pseudotsuga menziesii*) tends to be a dominant species, except along a narrow coastal strip dominated by Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*), and at subalpine elevations dominated by mountain hemlock (*Tsuga mertensiana*) and subalpine fir (*Abies lasiocarpa*).

The Fire Regime -- While one might expect that moist forests (sometimes >2500 mm annual precipitation) would rarely burn, a summer dry season is characteristic of this region, and annual precipitation drops to less than 500 mm in the southern portion of the region. The fire regime transitions from one of high-severity in the northern Cascades to a mixed-severity in the south as fire return intervals shorten. To the north, climate is the major driver of fire spread (Agee and Huff 1987), and large fires depend on previous winter drought, deep summer drought, ignition (lightning), and a dry, strong east wind (Agee 1991). The gradient of fire return intervals shown by field studies (Table 1) is also supported by a fire cycle model based on climate that shows about an order of magnitude increase in fire frequency from north to south in the region (Agee 1991). Regional synchroneity appears in these fire regimes, with more fire from the 1400s-1650 A.D., less fire from 1650-1800 A.D., and again more fire from 1801-1925 A.D. (Weisberg and Swanson 2003).

Large fires of 100,000-400,000 ha have occurred historically in the region (Fonda and Bliss 1969, Henderson et al. 1989, State of Oregon 1997). The most flammable conditions in these forests are in early successional stages (Agee and Huff 1987) so reburns are possible (Isaac 1940). Fonda and Bliss (1969) identified a series of large fires that covered the entire eastern Olympic Mountains of Washington, and Henderson et al. (1989) dated it to 1700 or 1701 and estimated their size at over 400,000 ha. Subsequently, a large subduction earthquake (9.0) on the Washington coast was dated to January 1700 (Atwater et al. 2005), which may have quake-thrown trees and created substantial dead fuel in a forest normally with few flashy dead fuels. Large fires in the Oregon Coast Range (100,000 ha +) occurred in 1848, 1853, and 1868 (State of Oregon 1997). The 175,000 ha Yacolt fire burned across southwestern Washington in 1902. This was followed by a large fire in the northern coast range of Oregon in 1933, ignited by a logging crew. The 96,000 ha Tillamook fire had a spectacular blowup, burning over 80,000 ha in just 20 hours. Major portions of this fire reburned in 1939, 1945, and 1951 due to snags catching fire and profuse cover of bracken fern, which is very flammable when cured (Isaac 1940). In 2002, the 200,000 ha Biscuit fire in southwestern Oregon burned for months across many successional stages (Raymond and Peterson 2005).

Forest Development Patterns **--** In northern portions of the region with high-severity fire regimes, fires tend to kill all the trees, either because the fire was a crown fire or the fire was hot enough to scorch the foliage of tall, old-growth trees (> 70 m) (Agee 1993). At

least two patterns of succession have been documented: 1) tree regeneration is immediate and 2) tree regeneration is delayed. These patterns may be due to the large scale of the event coupled with the availability of seed sources. In the Olympic Mountains, Huff (1995) found tree regeneration was rapid after fire burned 500, 180, 100, 20, and 1-3 year old stand, and that Douglas-fir establishment initially colonized the open landscapes created by the fire. After canopy closure, only western hemlock was able to establish, suggesting that the historical dominance of Douglas-fir over the region is testament to the importance of fire across the region. With most tree species in this region living 400-1000 years (Franklin and Dyrness 1973), only 1-2 fires per millennium are sufficient to maintain the dominance of Douglas-fir. Other evidence points to delays in forest regeneration, possibly due to lack of seed sources, shrub competition, or reburns (Franklin and Hemstrom 1981).

In the southern mixed-severity area of the Pacific Northwest, successional patterns are more complex (Weisberg 2004). Lower severity fires historically created variability in tree sizes and a greater component of shade-tolerant tree species. While Douglas-fir remains a dominant species, the diversity of other species is much higher than to the north. High burn severity patches in the landscape (often upper thirds of slopes and drier aspects; Taylor and Skinner 1998) typically regenerate to sprouting hardwood species or serotinous-coned conifers such as knobcone pine (*Pinus attentuata*). Lower severity patches typically have several age classes of Douglas-fir and associated regeneration can include Douglas-fir, hardwoods such as Pacific madrone (*Arbutus menziesii*) and canyon live oak (*Quercus chrysolepis*) , and shade-tolerant conifers such as white fir (*Abies concolor*) or grand fir (*Abies grandis*). Recent large scale fires such as the 1987 fires and Biscuit fire that have allowed the mixed-severity fire regime to again be emplaced upon the landscape thereby enhancing biodiversity (Martin and Sapsis 1991).

Managing landscapes -- Even with Douglas-fir as the primary tree species, the landscape effects of fire were quite unique between the mixed-severity and high-severity fire regimes. Patch edge was maximized in the mixed-severity fire regimes compared to the high-severity fire regime in the north or low-severity fire regimes like ponderosa pine to the east (Agee 1998), due to relatively small patches of differing fire severity. The patchy forest landscape created by mixed-severity fire regimes has maintained high fitness for northern spotted owls (*Strix caurina* var. *occidentalis*) (Franklin et al. 2000), preferable to either more homogeneous old growth or large patches of other vegetation types. Maintaining this patchiness using prescribed fire and wildland fire use will create major challenges for land managers in coming decades along with fragmentation from private forest land management (Spies et al. 1994).

To the north, large stand replacement patches of past centuries grew into old growth patches which form the basic concept of the Northwest Forest Plan with its late successional reserves (FEMAT 1993). Mean old growth in the Oregon Coast Range province (2,200,000 ha) appears to have averaged about 47%, but as the scale became more local (40,000 ha), old growth percentages ranged from 0-100% (Wimberly et al. 2000). A policy of fire exclusion might work as an ecosystem management strategy in these forests as northern spotted owls prefer large blocks of old growth for nesting and

roosting. However, when conditions are right, large fires are likely to occur and be relatively immune from firefighting efforts. Neither allowing all fires to burn nor suppressing all fires will be successful ecosystem management strategies across this complex region.

Northern Rockies

The topographic complexity of most northern Rocky Mountain landscapes, along with the convergence of maritime and continental climates, create diverse mosaics of vegetation communities and structures that are ultimately shaped by complex and dynamic fire regimes (Arno 1980, Habeck and Mutch 1973, Philpot 1990, Welner 1970). The juxtaposition of grasslands and xeric forests (e.g., ponderosa pine) with montane forests (e.g., Douglas-fir/western larch (*Larix occidentalis*) and nearby subalpine lodgepole pine/subalpine fir) requires that the entire landscape must be sufficiently dry to support the large fires of the past, and this occurs primarily during years of prolonged drought (Arno 1980, Barrett et al. 1991, Gruell 1983). These large fires were mostly started from lightning or Native American burning (Barrett and Arno 1982) during times of widespread regional drought. Kitzberger et al. (2007) found warm, dry springs are often precursors to large fire years.

Large fires were common on the northern Rocky Mountain landscape prior to 1980 AD. In regions of the upper Columbia River Basin, Barrett et al. (1997) found 35 large fire episodes occurred between 1540 and 1940 AD with most fire dates recorded across large regions of the interior Columbia River Basin. Most of these fire dates were recorded in xeric ecosystems with high fire frequency (e.g., ponderosa pine) and they burned 5-15% of the area. Barrett et al. (1997) also suggests that "major fires prior to 1900 burned more area than any fire years since". The fire years of 1910 and 1889 appear to be the largest in recent history (Koch 1942, Cohen and Miller 1978) but the fire years of 1869, 1856, 1846, 1833, and 1778 were also impressive in extent (Barrett et al. 1997). The difference between historical and contemporary large fires may be that today's large fires may burn areas that have deep duff layers, heavy woody fuel loadings, and thick canopies due to decades of fire exclusion that will certainly result in severe fires (Kolb et al. 1998), but it is unknown the extent of these same conditions prior to the fire exclusion era. Large fires have been increasing in recent years with 1988 and most years since 2000 having at least one large fire (Schoennagel et al. 2007).

Past large fires were important to regional ecology because they created large patches and complex mosaics that facilitated the regeneration and survival of many plant and animal species (Figure 1). For example, large patches created from regional fires ensured the continued presence of western larch because the mature trees had thick bark and high crowns that increased survival after high severity fires so it can disperse abundant seed in areas where all other trees were dead and seed sources were distant (Barrett et al. 1991, Davis 1980, Schmidt et al. 1976). Many large Rocky Mountain fires leave severity patterns that are quite diverse in shape and size which greatly effect subsequent post-fire ecological dynamics (Baker et al. 2007, Schoennagel et al. 2007). The intricate pattern of fire severity resulting from the large Yellowstone fires in 1988, for example, influenced

lodgepole pine regeneration (Turner et al. 2003), aspen and ungulate dynamics (Romme et al. 1995), and bird distributions dependent on stand replacement fires (Hutto 1995). This unique pattern will also affect future fire dynamics and landscape structure as successional development advances at disparate rates depending on the availability of propagules (e.g., level of serotiny, survival of rhizomes, tubers, and soil-borne seed, and sprouting potential), soil fertility, and post-fire climates (Turner et al. 1998, 1999).

Conventional wisdom holds that large fires tend to be more severe and therefore are undesirable (Laverty and Williams 2000, GAO 2002). To assess whether severity patterns from contemporary large fires were significantly different from small fires, digital burn severity layers were created for 11 large fires (>10,500 ha; see Figure 1 for an example) and 25 small fires (<3,300 ha) that occurred in the northern Rocky Mountains within the last 10 years (Table 2). These digital maps were generated from a classification of Landsat TM satellite imagery by deriving the differenced normalized burn ratio, and then linking to the composite burn index from approximately 1100 plots to obtain burn the severity classes in Table 2 (Key and Benson 2005). We then used FRAGSTATS (McGarigal and Marks 1995) to calculate important landscape metrics for each fire and summarized these metrics across large and small fires using statistical testing.

We found that a larger proportion of the burned landscape was, indeed, in the highest severity class for large fires as compared to that for small fires (high severity averaged 25 percent in large fires and 19 percent in small fires), but this difference was not significant ($p>0.05$, Table 2) and quite low (<25% of the area). In fact, there were few statistically significant differences in proportions of burned area for small and large fires across all burn severity classes. However, large fires were different from small fires in terms of landscape structure (Romme et al. 1998). Large fires tended to have larger patches (low patch density) that were more regular in shape (high landscape shape index) and had less edge than smaller fires (Table 2). However, the fractal dimension, contagion, and interspersion were nearly identical across small and large fires indicating that although the patches were larger, the patches tended to be adjacent to the lower severity classes creating diverse landscape mosaics.

There appears to be no indication that the frequency and severity of large fires in the northern Rocky Mountains have changed from historical fire regimes. This is partly because there has not been sufficient time (70+ years) to evaluate any temporal changes in long fire return interval ecosystems after the advent of fire exclusion policies. It is also because fighting fires during the weather and drought conditions that foster large fires tends to be difficult so fire suppression probably has the least effect on the size of large fires. Additionally, historical spatial distributions of fire severity in large fires are unknown because of the lack of spatially explicit field data, so it is difficult to compare today's fire severity patterns with historical patterns. Pre-settlement large fires tended to leave fire scarred trees on the landscape indicating low fire severities, but it is difficult to quantify the historical patch distribution of burn severity because evidence of most high severity burns is often lost. And last, large fire severity mosaics tend to be complex

(Figure 1, Table 2) so it would take measurements from many large fires to compute any statistically significant change in severity and frequency (Pratt et al. 2006).

Southern California

The Cedar Fire burned 110,600 ha of San Diego County shrublands in October 2003 and was the largest fire in the state since record keeping began about 90 years ago (Figure 2). Many hailed this as an anomalous event whose root cause was past land management practice coupled with an unusually long drought and proclaimed it would have disastrous repercussions for the biota. However, close examination of historical records and scientific studies fail to support these contentions.

The primary cause of the final size for the Cedar Fire was the severe weather event that coincided with an intentional human ignition in a remote portion of the backcountry. Like most all large wildfires in the western US, severe winds were a critical factor in the rapid spread of the fire. In southern California, large wind driven fires are a common feature of the landscape because of the annual foehn winds known locally as Santa Ana winds (Keeley and Fotheringham 2006). These occur every year and there may be anywhere from 10 to 50 or more such days a year. The worst of these are the autumn winds that come at the end of the annual drought, which is typically 4-6 months. These winds commonly exceed 100 km hr^{-1} with relative humidity less than 10% and they are capable of spreading fires more than 10,000 ha in an hour.

Many such fires have burned in the 20th century although none reached the ultimate size of the Cedar Fire. Nonetheless, if one considers a longer term view we see that larger fires have occurred historically. For example, during a Santa Ana wind event in the last week of September in 1889, southern California experienced a week of burning with one fire centered in Orange County that was reported as exceeding "over one hundred miles from north to south and is 10 to 18 miles in width." This 1889 fire would have been several times larger than the 2003 Cedar Fire, and like the Cedar Fire there were other large fires burning in the region at the same time (LA Times 1889). Thus, the Cedar Fire should be considered a 100-year event as opposed to an anomalous event. Other fire records such as marine sediments in the Santa Barbara Channel indicate a repeating cycle of large fire events for the region that extend the record back 500 years and indicate no change in the periodicity of such events (Mensing et al. 1999). The primary difference between the 1889 and 2003 events is the damage done to local communities. The 1889 fire is described "over $100,000 worth of pasturage and timber has been destroyed" whereas in the Cedar Fire 2,232 homes and 14 lives were lost (Keeley and Fotheringham 2006). Thus, the recent event was far more catastrophic in human terms, due primarily to the 100-fold increase in population density between 1889 and 2003 (http://www.census.gov/population/cencounts)

Past fire management practices, in particular fire exclusion and lack of effective fuels management, have been identified by scientists, managers and legislators in California as a key reason for the size of the Cedar Fire. Indeed, it has long been argued that large fire events such as this could be prevented by creation of landscape mosaics of fuels of

different ages (Philpot 1974, Minnich and Dizzani 1991). However, when these hypotheses are put to the test they have not been supported. Schoenberg et al. (2003), for example, found that after the first 2 decades of postfire recovery, there was no change in the risk of burning for Los Angeles County shrublands. Moritz et al. (2004) also analyzed the role of fuels throughout the southern and central coastal region and found that on 90% of the landscape fuel age was of minimal importance to fire hazard. The Cedar Fire is an excellent case in point as illustrated by the stand age map prior to the fire (Figure 2). Clearly, this landscape already represented a mosaic of fuel ages created by both wildfires and prescription burning and many patches were 20 years of age or less, yet the fire rapidly burned through age classes as young as 3 years. Maintaining a landscape dominated by even younger age classes is logistically problematical. Although 20 year old chaparral will burn readily when pushed by strong Santa Ana winds, it is difficult to re-burn chaparral younger than this under prescription burning conditions of wind speed and relative humidity (Keeley 2002).

These observations call into question the value of landscape scale prescription burning or other fuel modifications on these shrubland landscapes. One justification for continuing fuel modification work is that young fuels will sometimes act as barriers to fire spread under much more moderate weather conditions. However, such fires are seldom a threat to property or lives and thus a huge question mark remains as to whether they can be justified based on a cost/benefit ratio (Keeley 2005). Of course regardless of weather conditions fuel modified zones do reduce fire intensity and thus increase the defensible space for fire suppression forces. In a region experiencing urban sprawl, where fire fighting forces commonly are deployed in defensible postures during severe Santa Ana wind driven fires, the most strategic place for fuel modifications is directly at the wildland – urban interface.

Lastly, it was contended that the Cedar Fire was more intense than typical fires and, due to the intensity and the massive size of this fire, the recovery of these ecosystems was threatened. Crown-fire ecosystems such as chaparral are not affected by the size of this fire. This is because these ecosystems have endogenous mechanisms for recovery that include resprouting from basal burls and long-lived dormant seed banks that are stimulated to germinate by fire. Colonization is not important in their recovery and thus unlike many forest ecosystems, the landscape pattern of burning is largely unimportant in post-fire recovery. As for fire intensity, there is no evidence that fire intensity was greater in this fire than in many other chaparral fires, now or historically. In addition, these ecosystems are highly resilient to a range of fire intensities (Keeley et al. 2005).

Southwestern US

Large fires were historically extremely common in the southwestern U.S. (Arizona, New Mexico, and southern Utah and Colorado); a compilation of 63 fire history studies across the region showed that as many as two-thirds of the forests burned in synchrony in dry years such as 1748 (Swetnam and Baisan 2003). However, the critical difference between historical and modern fires is that past fires burned largely on the surface, while large modern fires such as the Rodeo-Chediski fire of 2002 burn primarily as crown fires.

The predominant forest type of the Southwest is ponderosa pine, generally as a monospecific forest type or together with Gambel oak. Southwestern ponderosa pine forests were historically characterized by frequent surface fire regimes (Swetnam and Baisan 2003). Disruption of the fire disturbance regime by livestock grazing, logging, and fire suppression triggered extensive tree regeneration (Cooper 1960), leading to dense forests that support canopy burning (Covington and Moore 1994, Fiedler et al. 2002). Recently, Westerling et al. (2006) argued that climate warming was directly associated with the increase in size and severity of western wildfire, but singled out the southwestern ponderosa pine forest as an example of the destructive convergence of warmer climate with historically unprecedented fuel levels.

In elevation zones surrounding the ponderosa pine, some southwestern forests were naturally characterized by severe fire. Across an elevational gradient from ponderosa pine through mixed conifer (ponderosa pine, Douglas-fir, white fir) to spruce-fir and aspen forest, historical fire regimes changed from frequent to infrequent surface fires, then to stand-replacing fires (Fulé et al. 2003). Since 1880, none of the high-elevation sites burned, increasing the continuity of dense fuels across the landscape (White and Vankat 1993). At low elevations in the Southwest, woodlands of piñon pines and junipers included varying fire regimes (Romme et al. 2003), but at least some were historically characterized by stand-replacing fires (Floyd et al. 2004).

Large crown fires in southwestern ponderosa pine forests have led to long-lasting changes in species dominance and habitats. A survey of severely burned landscapes found that only approximately a third recovered to support pine forests within the historical range of variability of forest structure (Savage and Mast 2005). Another third became “hyper-dense” forests, susceptible to crown fire again, and the remainder appeared to have converted to alternative stable states dominated by grass or brush (Savage and Mast 2005). In mixed pine-oak forests, severe burning favored regeneration by sprouting oaks at the expense of pines (Barton 2002). Invasive exotic species have capitalized on many large burns (Crawford et al. 2001, although invasives were notably limited following a severe fire in Grand Canyon National Park, perhaps because their seeds were not present in the relatively undisturbed, never-logged forest (Huisinga et al. 2005).

The Rodeo-Chediski fire complex in central Arizona in 2002, covering approximately 189,000 hectares, was by an order of magnitude the largest severe fire to date in the Southwest US. This landscape-scale fire may represent among the largest possible scales of fire in the region because it was essentially unaffected by almost all the attempted control actions and stopped only when it ran out of forest into the adjacent dry grasslands. Yet even under the conditions of extraordinary drought and sustained strong winds that supported the Rodeo-Chediski, those portions of the landscape with recent (< 11 yr old) treatments of prescribed fire or cutting + burning displayed primarily surface fire behavior (Finney et al. 2005, Strom 2005). These patches of living forest in a matrix of fire-killed vegetation are somewhat unique to the southwest US and provide three useful lessons for fire ecologists: (1) treated sites not only survived the worst-ever fire

conditions but also affected landscape-scale severe fire behavior by protecting adjacent untreated vegetation on the downwind side (Finney et al. 2005); (2) untreated forests suffered high mortality and vegetation simulation modeling projected long-term (100+ yr) dominance by oaks and other non-pine species, in contrast to continued pine dominance of treated sites (Strom and Fulé, in press); and (3) the survival of treated forests is strong empirical evidence of the effectiveness of ecological restoration and hazard fuel reduction efforts. The utility of such treatments in reducing fire severity had previously been shown in fire behavior simulations (e.g., Fulé et al. 2001) or stand-level post-fire comparisons (e.g., Pollet and Omi 2002).

Sagebrush Ecosystems

Sagebrush grasslands comprise the most widespread shrubland ecosystem in western North America (McArthur and Stevens 2004), subsequently, the consequences of variation and change in their fire regimes merit considerable interest. Here sagebrush grasslands are defined as those semi-arid landscapes co-dominated by one or more sagebrush taxa and by some combination of perennial grasses. Although the term, sagebrush, applies to a group of approximately 20 shrubby members of the genus *Artemisia* subgenus *Tridentatae* (Beetle 1960, McArthur 1979) sagebrush grasslands are primarily associated with the most widespread species, namely big sagebrush (*A. tridentata*), and to a lesser extent, silver (*A. cana*), black (*A. nova*), and low (*A. arbuscula*) sagebrushes. Taken as a whole, sagebrush grasslands cover a wide range in elevation (600-3,000 m) sharing ecotones with a variety of grassland, shrubland, woodland, and forest vegetation types (Wright and Bailey 1982, McArthur and Stevens 2004). Abundance and diversity of associated shrubs, grasses and forbs increase with increasing productivity potential and are correlated with soils and climate variables (Beetle 1960, Wright and Bailey 1982, McArthur and Stevens 2004).

Fire is the dominant disturbance force associated with historical sagebrush grasslands (Wright and Bailey 1982). Sagebrush plants are highly flammable and aboveground parts are quickly killed and consumed in most fires (Wright et al. 1979). Sagebrush regeneration is from seed with some exceptions (e.g., silver sagebrush sprouts from roots and crown; Wright et al. 1979). Because sagebrush seeds lack a mechanism for long-distance dispersal (Young and Evans 1989, Chambers 2000) and do not persist in soil seed banks (McDonough and Harniss 1974, Meyer and Monsen 1992), sagebrush recovery rates are long when fire intensity and size result in large uniform burns depleted of viable seed (Welch 2005). Without fire, many sagebrush grasslands are subject to encroachment and dominance by trees (Miller et al. 1999, Tausch 1999, Heyerdahl et al. 2006).

Historical Fire Regimes **--** Favorable years for large, spreading fires in sagebrush grasslands occur when hot dry summers follow wet spring conditions, indicating the importance of fine fuel (grass) accumulation and conditioning (Wright et al. 1979). Historical fire-free intervals are believed to have been shorter on more productive versus less productive landscapes due to greater average fuel production. Proximity to fire-prone

forest types (i.e., ponderosa pine and dry mixed-conifer) likely contributed to relatively short fire-free intervals on some landscapes.

Historical fire frequency has been estimated for a few sagebrush-grasslands landscapes using evidence collected from fire-scarred trees located near the forest/shrubland ecotone. Estimates of historical mean fire interval range from 12 years (Miller and Rose 1999) to 40 years (Houston 1973, Arno and Gruell 1983, Heyerdahl et al. 2006). Studies of post-fire succession suggest that mean fire-free periods of 30+ years are often needed for sagebrush recovery on productive sites (Harniss and Murray 1973, Humphrey 1984, Watts and Wambolt 1996, Wambolt et al. 1999, 2001) and much longer on less productive sites (Wright and Bailey 1982, West and Yorks 2002, Welch 2005). Subsequently, existing evidence suggest an historical mean fire interval of 35 to 80 years for productive landscapes and 100 to 200+ years for less productive sites maintained healthy sagebrush-grassland ecosystems.

Historical fire spatial patterns in sagebrush grasslands must be inferred due to difficulty in obtaining direct measures. Habitat requirements for big sagebrush-dependent species such as greater sage grouse (*Centrocercus urophasianus*) and pigmy rabbits (*Sylvilagus idahoensis*) suggest that large fires were locally rare (Crawford et al. 2004, Welch 2005). Conversely, precursor climate/fuel cycles favorable for extensive fire spread are common today and presumably were common historically. The potential for periodic large fires was greatest on productive, contiguous expanses of sagebrush grassland such as those that occurred from northern Nevada and eastern Oregon to western Wyoming. Short-term sagebrush recovery after large fires would have depended upon seedling establishment from un-burned, viable seed left in the seed bank or short-range seed dispersal from unburned islands and the burn perimeter. Both processes operate after modern fires and must have been important historically for between-fire recovery of sagebrush.

Modern Fire Regimes -- Sagebrush grassland fire frequency has changed relative to that of pre-Euro-American settlement due to the combined impacts of livestock grazing, fire suppression, exotic species introductions, and altered anthropogenic ignition patterns (Wright and Bailey 1982). Fire on many productive landscapes has been absent for 80-140 years. Encroachment by conifers is widespread and has resulted in partial to complete conversion to woodlands on favorable sites (Miller et al. 1999, Tausch 1999). Where encroachment is advanced, fuel loads are greatly increased and fuel type and structure are significantly altered from that of sagebrush grasslands. Woodland expansion also reduces landscape scale structural diversity (Tausch and Nowak 2000). Ultimately, woodland expansion pre-conditions landscapes for more extreme fire behavior and larger fires. Large fires during the past 20 years in piñon-juniper woodland – sagebrush grassland mosaics of Nevada and Utah confirm these expectations.

Fire intervals for many sagebrush grasslands of low to moderate productivity are currently 10 to 20 times shorter than what has been estimated for pre-settlement conditions (Whisenant 1990, Peters and Bunting 1994). This increase in fire is due primarily to the spread and dominance of the Eurasian annual, cheatgrass (*Bromus tectorum*), in sagebrush understories during the last century. This cool season grass cures

earlier than perennial grasses, effectively lengthening the fire season. The dense, somewhat continuous fine fuel matrix facilitates rapid fire spread and larger fires (Figure 3). Cheatgrass competitively excludes perennial seedlings effectively arresting post-fire succession. One effect of this cheatgrass-truncated succession is that multi-year series of adjacent fires have similar ecological impacts as single large fires. As a result, large continuous areas of sagebrush grasslands have been converted to annuals-dominated grasslands prone to short fire-free intervals and large fires (Whisenant 1990, Peters and Bunting 1994). Unburned sagebrush grasslands adjacent to these annual grasslands remain at high risk.

Although size of historic sagebrush-grassland fires is poorly understood, it is generally accepted that recent large fires were fueled by woodland-induced homogenization of landscapes and cheatgrass-dominated understories are outside the historical range of variation. These changes in fire regime and vegetation/fuel structure impact vast areas in the semi-arid western United States and are felt at all trophic levels. Effects are particularly harmful on landscapes where post-fire recovery is slowest. We can expect the trend for larger, more damaging fires in sagebrush grasslands to continue until aberrations in the fuel conditions that drive fire are corrected.

Piñon and juniper ecosystems

Piñon and juniper occupy over 30 million ha in the western United States (West 1999). In much of the Intermountain Region these trees are associated with sagebrush-steppe where they frequently form distinct woodlands adjacent to sagebrush communities or grow at varying densities within shrub-steppe communities (West 1999, Miller and Tausch 2001). Piñon and juniper occupy a wide variety of soils and the greater part grows in areas where annual precipitation is between 10 and 15 inches (Woodbury 1947, Gedney et al. 1999). The majority of present day stands of piñon and juniper can be separated into four general categories based on age and stand structure. Persistent woodlands are stands of trees that have been the dominant vegetation historically and currently are occupied primarily by trees well over 150 years old unless disturbed by fire or mechanical clearing. Persistent shrub-steppe savannas are communities usually dominated by low growing sagebrush species occupying shallow rocky soils and occupied by a low density of large trees. Persistent piñon and juniper savannas are communities typically dominated by warm season grasses and occupied by a low density of large trees. Piñon and juniper encroached sagebrush-steppe are communities that have been dominated by shrub-steppe vegetation historically but are currently in a transitional phase from shrub-steppe to woodland (Miller and Tausch 2001). Stand and age structure of the four categories are largely a function of climate, soils, topography, and the number of years between fire events. Fire regime and degree of anthropogenic influence widely differ among the four piñon and juniper categories. There are also communities that have been in a continual transitional state between shrub-steppe and woodlands that can be difficult to separate between persistent woodlands and tree encroached sagebrush-steppe.

Persistent woodlands -- Historic woodlands are commonly found on steep rocky slopes, shallow soils high in clay content, sedimentary soils, and soils high in sand, which limit

understory fuel loads (Burkhardt and Tisdale 1976, Tausch et al. 1981, Holmes et al. 1986, Burwell 1998, Miller and Rose 1995, 1999). Historic fire regimes in these woodlands, which accounted for the largest area impacted were typically infrequent (>200 years) high-severity stand replacement crown fires (Waichler et al. 2001, Baker and Shinneman 2004, Miller et al. 2005, Floyd et al. 2000). Spreading, low-intensity surface fires were probably rare in time and space across the majority of these woodlands as a result of sparse surface fuels (Baker and Shinneman 2004, Miller et al. 2000). However, small patchy burns within stands did occur but probably accounted for a small proportion of area impacted over time (Wangler and Minnich 1996, Whaichler et al. 2001).

Weather conditions under which these woodlands burned were typically severe (hot dry conditions with wind) often resulting in spatially large fires with limited spatial heterogeneity (Figure 3). However, variable topography and aspect strongly influence spatial complexity (Gruell et al. 1994). Increasing amounts of dead canopy foliage as a result of drought, insects, and disease also increased the potential for large fires to occur in closed woodlands. The potential for accumulation of fine fuels during several wet years is limited in most closed stands as a result of sparse understory vegetation (Cottam and Stewart 1940, West and Van Pelt 1986, Bates et al. 2000, Miller et al. 2000).

Little evidence suggests stand structure in closed-persistent piñon and juniper woodlands have changed in the past 150 years due to altered fire regimes (Baker and Shinneman 2004, Miller et al. 2005). In Mesa Verde, Floyd et al. (2000) concluded fire frequency during the past 50 years has not been greatly different from the fire regime in the 1800s. Recent large stand replacement events in persistent closed woodlands are probably not out of the range of historic variability for closed persistent woodlands. However, detecting change in the past 100 plus years is difficult for historic fire return intervals measured in centuries. Longer fire seasons and higher summer temperatures during the past decades (Westerling et al. 2006) and increased canopy cover and foliage biomass resulting from elevated CO_2 levels (Knapp and Soulé 1996, Knapp et al. 2001, Soulé et al. 2004), will likely increase the probability that these historic woodlands will burn. This could result in the decline of these old-growth closed persistent woodlands.

A large proportion of persistent woodlands present at the time of Eurasian settlement established during the Little Ice Age (approximately 1300-1850). So it is difficult to determine the effects of current climate on the development of historic woodlands following disturbance. In addition to climate, woodland succession is largely dependent on the understory composition prior to the fire and fire severity (Erdman 1970, Barney and Frischknecht 1974, Everett and Ward 1984, Koniak 1985, Wangler and Minnich 1996). In the absence of weed encroachment, the response of native biota and successional trajectories appear to have changed little across this vegetation group, although we have no information on the long term development of the trees. In the presence of exotic plants such as cheatgrass (*Bromus tectorum*), successional trajectories can dramatically change following woodland fires where native understory species are depleted due to high fire severity, over-grazing, and or competition from the trees. The contiguous cover of exotic annuals can result in repeated fires, which limits the

establishment of native species creating a new steady-state of vegetation (Tausch 1999b). The decline of these persistent woodlands would have a negative impact on landscape diversity and cavity nesting species. Densities of cavity nesting birds were significantly greater in old-growth woodlands compared to shrub-steppe, tree encroached shrub-steppe, and recent burns resulting in grass dominated communities (Reinkensmeyer et al. 2007).

Persistent tree-shrub savannas **--** Piñon and juniper-shrub savannas extend across large areas in the northern Great Basin. These communities are characterized by tree canopies typically <10%, low growing sagebrush species, particularly low and black sagebrush (*Artemisia arbuscula, A. nova*), and shallow-rocky soils. Infrequent surface fires are limited by low fuel abundance and continuity. Although little information is available on fire regimes in this vegetation category, two studies reported fire return intervals between 90 and 150 years (Young and Evans 1981, Miller and Rose 1999). Fire regimes within this category were probably infrequent (100-200 years), mixed severity fires. Fuel accumulation over several wet years is important in preconditioning this vegetation group to burn. In central Oregon, two relatively extensive fires burned across a low sage-brush community in 1717 and 1855 (Miller and Rose 1999). Both years were preceded by two or more, wetter than average years based on tree ring growth. The fire in 1855 killed about a fourth of the mature trees.

There is no evidence that fire regimes have markedly changed in this vegetation group. Past and current fuels have naturally limited fires to relatively long fire free intervals. Tree density and canopy cover, however, have increased in some areas within piñon-juniper savannas. The primary risk of these sites shifting to another steady state following fire is where native species have been replaced by introduce annuals, usually as a result of over-grazing. On the heavy textured soils Medusa-head (*Taeniatherum capute-madusae*) is of greatest concern. The majority of native herbaceous understory species respond quickly following fire. However, low sagebrush may take decades to reestablish and tree establishment and growth are slower than occurs on deeper and relatively more well drained soils occupied by the big sagebrush subspecies (*Artemisia tridentata*).

Persistent piñon and juniper savannas **--**Very little information is available on historic piñon-juniper savannas, which probably accounted for a small portion of the total piñon and juniper stands in the West. Contiguous fine fuels resulted in frequent low-intensity surface fires which supported the persistent coverage of herbaceous vegetation and scattered trees (West 1999). These tree savannas were most common in the Southwest and probably heavily impacted by grazing in the 1800s. The reduction in fine fuels altered the fire regime to infrequent fires resulting in the increase in woody vegetation.

Piñon and juniper encroached sagebrush-steppe **--** Historic fire regimes for the sagebrush ecosystem are covered in the previous section. However, extensive encroachment of shrub-steppe communities by conifers throughout the Intermountain Region makes it difficult to separate shrub-steppe from woodlands. Where piñon and juniper seed sources are available, fire return intervals of less than 50-80 years is

probably necessary to limit tree encroachment and maintain shrub-steppe communities in the deeper more productive soils occupied by big sagebrush subspecies. Growth of piñon and juniper is usually relatively slow during the first 45-50 years (Miller and Tausch 2001). In mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), Miller and Rose (1999) reported 45-50 years were required for the majority of juniper trees to reach a 3m height. Trees <3m in height are easily killed by fire (Jameson 1962, Dwyer and Pieper 1967, Burkhardt and Tisdale 1976, Bunting 1984). Following fire, a minimum of 80 years is required for trees to begin dominating a site, resulting in reduced understory fuels particularly shrubs, which are important to carry the fire from the ground to the crown (Barney and Frischknecht 1974, Johnson and Miller 2006). Fire return intervals exceeding 50-80 years would allow trees to approach a more fire resistant size and a decline in surface and ladder fuels.

Several studies suggest that large fires in sage-brush steppe occurred in years following one or more wet years resulting in fine fuel accumulation (Baisan and Swetnam 1990, 1997; Miller and Rose 1999). Sagebrush steppe communities, particularly mountain big sagebrush, can occur on landscapes of widely varying topography and soils resulting in heterogeneous fuels. The majority of historic fires were small with less frequent large fires. However, as trees begin to dominate a site, suppression of understory species reduces the susceptibility of the site to fire. As these woodlands approach the late stages of development and canopies become dense, stands become more susceptible to severe stand replacement fires during periods of drought and sufficient wind velocities (Tausch 1999a, b). These stand replacement fires were probably rare prior to European settlement. In Texas, woodlands with tree canopy cover exceeding 35% were capable of supporting a crown fire (Bryant et al. 1983). Recent surveys in Nevada, eastern California, southeastern Oregon, and southwestern Idaho indicate closed piñon-juniper stands currently represent 10 to 50% of the total woodland area. The area of closed stands will likely more than double in the next 40 to 50 years (USDA Humboldt Toyiabe National Forest 1999, Johnson and Miller 2006).

Response of native species following fire is influenced by climate, state of woodland succession, species composition, seed pools, and fire severity (Erdman 1970, Barney and Frischknecht 1974, Everett and Ward 1984). The rate of tree reinvasion has been associated to piñon and juniper seed pools and state of woodland succession at the time of disturbance (Barney and Frischknecht 1974, Burkhardt and Tisdale 1976). Exotic plants such as cheatgrass can respond rapidly following woodland fires where native understory species are depleted due to high fire severity, over-grazing, and or competition from the trees. The contiguous cover of exotic annuals can result in repeated fires, which limits the establishment of native species creating a new steady-state of vegetation (Tausch 1999).

There are communities that have historically been in a continual transitional state between big sagebrush-steppe and woodland. These communities are probably a function of intermediate fire return intervals (80-200 years) that allow shrubs to persist for a major portion of the transitional period and woodlands to develop but never fully mature into persistent stands. In northeastern California, this vegetation group occupied the relatively

more arid mountain big sagebrush sites characterized by limited surface fuels (Miller and Heyerdahl in progress). Stand replacement disturbance events occurred in the early 1700s and 1856, the last following several wet years following fuel accumulation. Fire suppression and reduction of fine fuels due to livestock grazing probably have shifted a larger proportion of this to group towards woodland development. Response of key biota following fire under current conditions will be influenced by stage of woodland development, composition of native and exotic plants, soils, fire severity, and climate.

Fire regimes in persistent woodlands and shrub-steppe savannas across the Intermountain Region are still probably within the historic range of variability. In the absence of exotic plants, the successional trajectory following fire appears to shift from native herbaceous to shrubland and in woodlands to piñon or juniper dominated communities. However, global warming, increased atmospheric CO_2, and the presence of exotic plant species threatens the decline of these old-growth woodlands, shifting them to introduced herblands. In contrast, fire regimes across much of the piñon and juniper savannas and encroached shrub-steppe have changed considerably since the late 1800s. In the absence of fire conifers will continue to establish in big sagebrush communities (Heyerdahl et al. 2006, Miller and Rose 1999, Miller and Tausch 2001). As woodlands approach the late stages of development and understory vegetation becomes sparse they become more susceptible to severe stand replacement fires and the invasion of exotic plants. The replacement of native understory species by annual and biannual exotic plants can result in increased fire occurrence, size, and decrease fire spatial complexity.

Great Lakes conifer-hardwood forests

Fire regimes of mixed conifer-hardwood forests within the northern Great Lakes Region are also complex and heterogeneous, and depend on local interactions between climate, physiography, vegetation, landscape context, and human factors (Heinselman 1973, Cardille et al. 2001, Cleland et al. 2004, Schulte et al. 2005). Dry conditions conducive to fire occur on an annual basis in the upper Great Lakes region, and are usually correlated with the prevalence of warm dry southwesterly winds (Lorimer and Gough 1988). Extremely dry conditions occur on a decadal basis (Lorimer and Gough 1988). The fire season spans April to October and, across much of the region, peak fire incidence is prior to leaf out in the spring or in the fall (Haines et al. 1975, Lorimer and Gough 1988). The fire season can span the whole summer, however, where climate, physiography, and vegetation synergistically interact to form extremely fire-prone systems such as the Boundary Waters area of northeastern Minnesota and the Mio Outwash Plains of the northern Lower Peninsula of Michigan (Haines et al. 1975), which tend to experience the largest fires in the region (Heinselman 1973, Simard and Blank 1982).

The pre-Euro-American fire regime included both large, intense stand-replacing fires with variable frequencies and high frequency, low intensity surface fires (Heinselman 1973, Cleland et al. 2004, Schulte et al. 2005). Large conflagrations primarily occurred on or nearby locations with shallow soils and on glacial outwash plains. These drought-prone landforms also influenced fire frequencies within surrounding landscapes; fires

ignited on droughty landforms spread to neighboring, more mesic landforms, affecting vegetation composition and structure (Cleland et al. 2004, Schulte and Mladenoff 2005). Although large and severe fires were common on these locations, light to moderate surface fire were also prevalent and important in structuring vegetation in some historical forests because these frequent (1-2 times per decade), low intensity surface fires maintained savannas, barrens, and pine forests (Heinselman 1973, Frelich and Lorimer 1991, Cleland et al. 2004, Schulte and Mladenoff 2005).

The most infamous large fires in the northern Great Lakes region occurred during the Euro-American settlement period. According to Pyne (1997), "The first railroad entered northern Wisconsin in 1870. It was succeeded a year latter by the worst fire disaster in American history. For about 60 years this pattern was repeated throughout the northwoods." Some of these infamous fires include the 1871 Peshtigo Fire (518,000 ha), the 1881 Thumb Fire (41,000 ha), the 1894 Hinckley Fire (65,000 ha), the 1894 Phillips Fire (39,000 ha), the 1908 Metz Fire, the 1910 Baudette Fire (120,000 ha), the 1911 Oscoda-Ausable Fire, and the 1918 Cloquet Fire (101,000 ha) (Mitchell and Robson 1950, Haines and Sando 1969, Lorimer and Gough 1988), many of which claimed many human lives in addition to consuming forest (Haines and Sando 1969). The substantial buildup of logging slash and the careless treatment of fire (e.g., unchecked fires started by farmers for land clearing, railroads, and campers) at the time primed the environment for forest fire (Pyne 1997, Pernin 1999); when these conditions paired with the severe drought that periodically occurs in the northern Great Lakes region, conflagrations were the result (Lorimer and Gough 1988). Many settlement period fires burned several times more area than the largest previously-recorded fires in the region. The Peshtigo fire, for example, covered an area almost three times as large as the largest known natural fire (the 1864 fire recorded in the BWCA at 180,000 ha) (Heinselman 1973).

The number and size of fires dramatically dropped as fire control became effective in the early part of the 20th Century and has been low ever since (Heinselman 1973, Cleland et al. 2004). For example, 232,000 hectares burned annually in Michigan between 1910 and 1925; this average decreased to just 11,000 ha between 1939 and 1948 (Mitchell and Robson 1950). An increase in fire activity is not expected at present because the regional forest has "switched" to a new, less fire-prone state (Frelich and Reich 1995; Sturtevant et al. 2004). This switch occurred with the expansion of broad-leaved deciduous species such as sugar maple (*Acer saccharum*), a shade-tolerant, competitively-dominant tree in the region, into historically pine- (*Pinus banksiana*, *P. resinosa*, *P. strobus*) and oak-dominated (*Quercus macrocarpa*, *Q. rubra*, *Q. ellipsoidalis*) systems. Once established, shade-tolerant, deciduous species tend to inhibit fire ignition and spread (Sturtevant et al. 2004). The interaction between fire suppression, fuel loads, and wildfire in the northern Great Lakes region is, thus, quite different from other locations, such as western U.S. Although fire is still a part of northern Great Lakes forests today, present landscapes are so fragmented by human land use (e.g., forestry, agriculture, settlement) that it is difficult to tease apart the effect of the natural dynamic from the human one; rather human factors strongly interact with abiotic and biotic conditions to determine present fire dynamics (Cardille et al. 2001, Cleland et al. 2004, Sturtevant et al. 2004).

Large fires are presently scattered across the region, but are more prevalent within northwestern Minnesota, central Minnesota, and the north central portion of the Michigan's Lower Peninsula (Cardille et al. 2001). A few very large fires on record for contemporary times include the 1971 Little Sioux Fire in Minnesota (6,000 ha), the 1976 Seney Fire in Michigan (24,000 ha), and 1980 Mack Lake Fire in Michigan (16,000 ha) (Heinselman 1973, Pyne 1997). Although the broad scale distribution of fire is similar to the historical one, most fires and most large fires today occur in the Aspen-Birch forest type (Cardille and Ventura 2001); whereas, historically, most fires occurred in conifer, mixed conifer-hardwood, or oak systems (Heinselman 1973, Zhang et al. 1999, Cleland et al. 2004, Schulte et al. 2005). At a finer scale, the occurrence of large fires is highest in locations proximal to nonforest land cover, with low stream densities, and with low road densities (Cardille et al. 2001). Large fires tend to occur in areas that are distant from cities, close to nonforest land cover, and with low lake densities (Cardille et al. 2001). Because most fires are the result of human ignitions, Cardille et al. (2001) expect large fires started in more remote locations have a lower probability of being reported while small, hence allowing time for the fires to increase in size.

Are large fires a benefit or catastrophe in the northern Great Lakes region today? Given the sweeping differences between pre-settlement and present fire regimes, a major ecological concern is how any type of fire and its benefits might be reintroduced to the current, more fully humanized landscape (Radeloff et al. 1999, Schulte et al. In review). Restoring pine and oak systems in the region will require fire. These historically-dominant species need the high light and bare mineral soil conditions created by fire for successful germination (Burns and Honkala 1990). Fire also allows them to maintain their competitive edge against more shade-tolerant, broad-leaved deciduous species (e.g., *A. saccharum*, *A. rubrum*, *Tilia americana*), especially on higher quality sites (Will-Wolf and Stearns 1999, Sturtevant et al. 2004). Those systems requiring the most frequent fire (1-2 times/decade), including pine barrens, pine savanna, and oak savanna systems, are among those most threatened habitats globally today, and are home to many threatened and endangered species, including the Kirtland's Warbler (*Dendroica kirtlandii*), Sharp-tailed Grouse (*Tympanuchus phasianellus*), the Karner Blue butterfly (*Lycaeides melissa samuelis*), and several other butterfly and moth populations (Pregitzer and Saunders 1999, Will-Wolf and Stearns 1999). Restoring and maintaining these systems and species into the future will require active and concerted effort toward fire reintroduction and management.

Summary

In general, it appears that large fires were common on historical landscapes of the US except for sagebrush ecosystems, and they will continue to occur on contemporary landscapes (Table 3). In many US ecosystems, large fires were an important ecosystem disturbance and many plant and animal species have adapted to survive and thrive after these extensive events. However, large fire characteristics and effects appear to differ across most ecosystems presented here. Sagebrush ecosystems are currently experiencing larger, more severe, and more frequent fires compared to historical conditions due to exotic cheatgrass invasions. Similarly, large fires in southwest

ponderosa pine forest historically created a mixed severity mosaic dominated by non-lethal surface fires while today's large fires are mostly high severity crown fires. This is quite different from the other presented ecosystems where historical and current large fire effects are comparable (Table 3). While large fires play an important role in landscape fire ecology for most regions, their importance is much less in the dry Piñon juniper forests and sagebrush grasslands. We emphasize that there is limited data to critically evaluate any changes in historical large fire effects because most large fires have removed evidence of past severity patterns and large fire return intervals tend to be long (Turner et al. 1998) making it difficult to determine the effects of land management (i.e., fire exclusion, resource extraction) that occurred over a relatively shorter time. Future fire management should recognize the importance of these megafires to regional ecology.

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References

Agee JK (1990) The historical role of fire in Pacific Northwest forests. In 'Natural and prescribed fire in Pacific Northwest forests' (Eds JD Walstad, SR Radosevich, DV Sandberg) pp. 25-39 (Oregon State University Press: Corvallis OR).

Agee JK (1991) Fire history of Douglas-fir forests in the Pacific Northwest. In: 'Wildlife and vegetation of Douglas-fir forests in the Pacific Northwest' (Eds L Ruggiero, K Aubry, M Huff) pp 25-33. (USDA Forest Service Pacific Northwest Research Station Report PNW-GTR-285).

Agee JK (1993) Fire ecology of Pacific Northwest forests (Island Press: Washington, DC)

Agee JK (1998) The landscape ecology of western forest fire regimes. *Northwest Science* **72** (special issue), 24-34.

Agee JK, Huff MH (1987) Fuel succession in a western hemlock/Douglas-fir forest. *Canadian Journal of Forest Research* **17**, 697-704.

Agee JK, Krusemark F (2001) Forest fire history of the Bull Run, Oregon. *Northwest Science* **75**, 292-306.

Albert, DA (1995) 'Regional landscape ecosystems of Michigan, Minnesota, and Wisconsin: a working map and classification' USDA Forest Service North Central Research Station General Technical Report NC-178 (St. Paul, Minnesota, USA)

Arno SF (1980) Forest fire history of the northern Rockies. *Journal of Forestry* **78**, 460-465.

Arno, SF (1998) 'Fire disturbance and associated impacts on forest values: Some implications of fire exclusion' Pages 1-3 *in* Proceedings from the Western Forest Insect Work Conference. USDA Forest Service, Rocky Mountain Research Station (Jackson, WY)

Arno, SF and J K Brown. (1991) Overcoming the paradox in managing wildland fire. Western Wildlands .

Arno, SF, Gruell, GE (1983) Fire history at the forest-grassland ecotone in southwestern Montana. *Journal of Range Management* **36**, 332-336.

Atwater B, Musumi-Rokkaku S, Satake K, Ueda K, Yamaguchi DK (2005) The orphan tsunami of 1700 (United States Geological Survey: Reston, VA and University of Washington Press: Seattle, WA)

Baker, WL (1992) Effects of settlement and fire suppression on landscape structure. *Ecology* **73**:1879-1887.

Baker, WL (2003) 'Fires and climate in forested landscapes of the US Rocky Mountains' Pages 120-157 *in* T. T. Veblen, W. L. Baker, G. Montenegro, and T. W. Swetnam, editors. Fire and climatic change in temperate ecosystems of the western Americas (Springer-Verlag, New York, New York, USA)

Baker, WL and DJ Shinneman (2004) Fire and restoration of piñon and juniper woodlands in western United States: a review. *Forest Ecology and Management* 189:1-21.

Baker WL, Veblen TT, Sherriff RL (2007) Fire, fuels, and restoration of ponderosa pine-Douglas-fir forests in the Rocky Mountains, USA. *Journal of Biogeography* **34**, 251-269

Baisan, CH and TW Swetnam (1990) Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, USA. *Canadian Journal of Forest Research* 20: 1559-1569.

Baisan, CH and TW Swetnam (1997) 'Interactions of fire regimes and land use in the Central Rio Grande Valley' USDA, Forest Service, Rocky Mountain Forest and Range Research Paper RM-RP-330 (Fort Collins, CO)

Barney, MA and NC Frischknecht (1974) Vegetation changes following fire in the pinyon-juniper type of west-central Utah. *Journal of Range Management* 27:91-96

Barrett SW, Arno SF (1982) Indian fires as an ecological influence in the northern Rockies. *Journal of Forestry*, 647-651.

Barrett, SW, SF Arno and JP Menakis (1997) 'Fire episodes in the inland northwest (1540-1940) based on fire history data' USDA Forest Service Intermountain Research Station General Technical Report INT-GTR-370 (Ogden, UT)

Barton AM (2002) Intense wildfire in southeastern Arizona: transformation of a Madrean oak-pine forest to oak woodland. *Forest Ecology and Management* 165, 205-212.

Bates J, Miller RF, Svejcar TS. 2000. The effects of cutting *Juniperus occidentalis* Hook. on understory diversity, cover, and biomass. *Journal of Range Management* **53,** 119-126.

Bessie WC and EA Johnson (1995) The relative importance of fuels and weather on fire behaviour in subalpine forests. *Ecology* **76**, 747-762.

Beetle AA (1960) 'A study of sagebrush. The section *Tridentatae* of *Artemisia'* Agricultural Experiment Station Bull 368. University of Wyoming (Laramie, WY). 83 p.

Bridge SRJ, Miyanishi K, Johnson EA (2005) A critical evaluation of fire suppression effects in the boreal forest of Ontario. *Forest Science* **51**:41-50.

Brown JK (1985) 'The "unnatural fuel buildup" issue' Pages 127-128 *in* J. E. Lotan, B. M. Kilgore, W. C. Fischer, and R. W. Mutch, editors. Symposium and Workshop

on Wilderness Fire. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station (Missoula, Montana)
Bryant FC, GK Launchaugh and BH Koerth (1983) Controlling mature ash juniper in Texas with crown fires. *Journal of Range Management* 22:264-270.
Bunting SC (1984) 'Prescribed burning of live standing western juniper and post-burning succession' Pages 69-73. In*:* T.E. Bedell (compiler), Oregon State University Extension Service Proceedings Western juniper short course. October 15-16, (Bend, OR)
Burkhardt JW, EW Tisdale (1969) Nature and successional status of western juniper vegetation in Idaho. *Journal of Range Management* **22**, 264-270.
Burns RM, Honkala BH (1990) 'Silvics of North America. Volumes 1 and 2' USDA Forest Service Agriculture Handbook 654 (Washington, D. C., USA)
Burwell T (1998) Successional patterns of the lower montane treeline, eastern California. *Madroño* **45**, 12-16.
Butry DT (2001) What is the price of catastrophic wildfire? *Journal of Forestry* **99**, 9-17.
Calkin DE, Gebert KM, Jones JG, Neilson RP (2005) Forest Service large fire area burned and suppression expenditure trends, 1970-2002. *Journal of Forestry* **November 2005**, 179-183.
Cardille JA, Ventura SJ (2000) Occurrence of wildfire in the northern Great Lakes region: effects of land cover and land ownership assessed at multiple scales. *International Journal of Wildland Fire* **10**, 145-154.
Cardille JA, Ventura SJ, Turner MG (2001) Environmental and social factors influencing wildfires in the upper Midwest, United States. *Ecological Applications* **11**, 111-127.
Chambers JC (2000) Seed movements and seedling fates in disturbed sagebrush steppe ecosystems: implications for restoration. *Ecological Applications* **10**, 1400-1413.
Cleland DT, Crow TR, Saunders SC, Dickman DI, Maclean AL, Jordon JK, Watson AK, Sloan AM, Brosofske KD (2004) Characterizing historical and modern fire regimes in Michigan (USA): A landscape ecosystem approach. *Landscape Ecology* **19**, 311-325.
Cohen S, Miller D (1978) The Big Burn -- The Northwest's forest fire of 1910 (Pictoral Histories Publishing Co., Missoula, MT., USA)
Cooper CF (1960) Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. *Ecology* **42**, 493-499.
Cottam WP, Stewart G (1940) Plant succession as a result of grazing and of meadow desiccation by erosion since settlement in 1862. *Journal of Forestry* **38**, 613-626.
Courtney SP, Blakesley JA, Bigley RE, Cody ML, Dumbacher JP, Fleisher RC, Franklin AB, Franklin JF, Gutierrez RJ, Marzluff JM, Sztukowski L (2004) Scientific evaluation of the status of the Northern Spotted Owl. Sustainable Ecosystems Institute (Portland, OR)
Crawford JA, Wahren C-HA, Kyle S, Moir WH (2001) Responses of exotic plant species to fires in *Pinus ponderosa* forests in northern Arizona. *Journal of Vegetation Science* **12**, 261-268.
Crawford, JA, Olson RA, West NE, Mosley JC, Schroeder MA, Whitson TD, Miller RF, Gregg MA, Boyd CS (2004) Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* **57**, 2-19.

Cui W, Perera, A (2006) Forest fire size distribution in North American boreal forests. Forest Research Information Paper No. 163, Ontario Forest Research Institute, Ontario Ministry of Natural Resources (Sault Ste. Marie, Ontario, Canada)

Daniel TC, Carroll MS, Moseley C, Raish C (2007) (Eds) 'People, fire, and forests: a synthesis of wildfire social science.' (Oregon State University Press: Corvallis, OR), 226 pages.

Davis KM (1980) Fire history of a western larch/ Douglas-fir forest type in northwestern Montana. In 'Proceedings of the fire history workshop'. Tucson, Arizona. (Eds MA Stokes and TC J.H. Dietrich) pp. 69-74. (Rocky Mountain Forest and Range Experiment Station, Forest Service, USDA)

Dwyer DD, Pieper RD (1967) Fire effects on blue grama-pinyon rangeland in New Mexico. *Journal of Range Management* **20**, 359-362.

Eberhart KE, Woodward PM (1987) Distribution of residual vegetation associated with large fires in Alberta. *Canadian Journal of Forest Research* **17**, 1207-1212.

Erdman JA (1970) Pinyon-juniper succession after natural fires on residual soils of Mesa Verde, Colorado. *Brigham Young University Science Bulletin, Biological Series* **11**, 1-26.

Everett RL, Ward K (1984) Early plant succession in pinyon-juniper controlled burns. *Northwest Science* **58**, 57-68.

Fahnestock GR, Agee JK (1983) Biomass consumption and smoke production by prehistoric and modern forest fires in western Washington. *Journal of Forestry* **81**, 653-657.

Ferry GW, Clark RG, Montgomery RE, Mutch RW, Leenhouts WP, Zimmerman GT (1995) 'Altered fire regimes within fire-adapted ecosystems' U.S Department of the Interior --National Biological Service (Washington, DC)

Fiedler CE, Keegan III CE, Robertson SH, Morgan TA, Woodall CW, Chmelik JT (2002) A strategic assessment of fire hazard in New Mexico. Final report to the Joint Fire Science Program (Boise, ID)

Finney MA, McHugh CW, Grenfell IC (2005) Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* **35**, 1714-1722.

Floyd LM, Hanna DD, Romme WH (2000) Fire history and vegetation pattern in Mesa Verde national Park, Colorado, USA. *Ecological Applications* **10**, 1666-1680.

Floyd ML, Hanna DD, Romme WH (2004) Historical and recent fire regimes in piñon-juniper woodlands on Mesa Verde, Colorado, USA. *Forest Ecology and Management* **198**, 269-289.

Forest Ecosystem Management Assessment Team (FEMAT) (1993) Forest Ecosystem: An ecological, economic, and social assessment. US Department of Agriculture, US Department of Interior, and others (Portland, OR)

Fonda RW, Bliss LC (1969) Forest vegetation of the montane and subalpine zones, Olympic Mountains, Washington. *Ecological Monographs* **39**, 271-301.

Franklin AB, Anderson DR, Gutierrez RJ, Burnham KP (2000). Climate, habitat quality, and fitness in northern spotted owl populations in northwestern California. *Ecological Monographs* **70**, 539-590.

Franklin JF, Dyrness CT (1973) Natural vegetation of Oregon and Washington. USDA Forest Service Pacific Northwest Research Station General Technical Report PNW-8 (Portland, OR)

Franklin JF, Hemstrom M (1981) Aspects of succession in the coniferous forests of the Pacific Northwest. In ‘Forest succession: concepts and application’ (Eds DC West, HH Shugart, DB Botkin) pp. 222-229 (Springer-Verlag, NY)

Franklin JF, Spies TA, Van Pelt R, Carey AB, Thornburgh DA, Berg DR, Lindenmayer DB, Harmon ME, Keeton WS, Shaw DC, Bible K, Chen J (2002) Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecology and Management* **155**, 399-423.

Frelich LE (2002) Forest dynamics and disturbance regimes: studies from temperate evergreen-deciduous forests (Cambridge University Press, Cambridge, U.K.)

Frelich LE., Lorimer CG (1991) Natural disturbance regimes in hemlock-hardwood forests of the Upper Great Lakes region. *Ecological Monographs* **61**,145-164.

Frelich LE, Reich PB (1995) Neighborhood effects, disturbance, and succession in forests of the western Great Lakes region. *Ecoscience* 2, 148-158.

Fulé PZ, Waltz AEM, Covington WW, Heinlein TA (2001) Measuring forest restoration effectiveness in hazardous fuels reduction. *Journal of Forestry* **99(11**), 24-29.

Fulé PZ, Crouse JE, Heinlein TA, Moore MM, Covington WW, Verkamp G (2003) Mixed-Severity Fire Regime in a High-Elevation Forest: Grand Canyon, Arizona. *Landscape Ecology* **18**, 465-486.

Fulé PZ, Cocke AE, Heinlein TA, Covington WW (2004) Effects of an intense prescribed forest fire: Is it ecological restoration. *Restoration Ecology* **12**, 220-230.

GAO (2002) Severe wildland fires: leadership and accountability needed to reduce risks to communities and resources. Report to Congressional Requesters GAO-02-259,)United States General Accounting Office, Washington DC)

Gardner R, Hargrove, WR, Turner MG (1997) Effects of scale-dependent processes on predicting patterns of forest fires. *Landscape Ecology*

Gedney DR, Azuma DL, Bolsinger CL, McKay N (1999) ‘Western juniper in eastern Oregon’ USDA Forest Service Pacific Northwest Research Station General Technical Report PNW-GTR-464.

Gruel GE, Eddleman LH, Jaindl R (1994) ‘Fire history of the pinyon-juniper woodlands of Great Basin National park’ U.S. Department of Interior, National Park Service, Pacific Northwest Region Technical Report NPS/PNROSU/NRTR-94/01 (Seattle, WA)

Habeck JR, Mutch RW (1973) Fire-dependant forests in the northern Rocky Mountains. *Quaternary Research* **3**, 408-424.

Haines DA, Sando RW (1969) ‘Climatic conditions preceding historically great fires in the north central region’ USDA Forest Service, North Central Forest Experiment Station Research paper NC-34 (St. Paul, Minnesota, USA) 19 pp.

Haines DA, Johnson VJ, Main WA (1975) ‘Wildfire atlas of the northeastern and northcentral states’ USDA Forest Service North Central Forest Experiment Station General Technical Report NC-16 (St. Paul, Minnesota, USA) 25 pp.

Harniss RO, Murray RB (1973) 30 years of vegetal change following burning of sagebrush-grass range. *Journal of Rangeland Management* **26**, 322-325.

Heinselman ML (1973) Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. *Quaternary Research* **3**, 320-382.

Hutto RL (1995) Composition of bird communities following stand-replacement fires in northern Rocky Mountains (USA) conifer forests. *Conservation Biology* **9**, 1041-1058.

Lorimer CG, Gough WR (1988) Frequency of drought and severe fire weather in northeastern Wisconsin. *Journal of Environmental Management* **26**, 203-219.

Helgerson O (1988) Historic fire year for Oregon and California. *Oregon State University Forestry Intensified Research Report* **9(4)**, 2-4.

Hemstrom M, Franklin JF (1982) Fire and other disturbances of the forests in Mount Rainier National Park. *Quaternary Research* 18, 32-51.

Henderson JA, Peter DH, Lesher RD, Shaw DC (1989) Forested plant associations of the Olympic National Forest. USDA Forest Service Pacific Northwest Region ECOL Technical Paper 001-88. (Portland, OR)

Heyerdahl EK, Miller RF, Parsons RA (2006) History of fire and Douglas-fir establishment in a savanna and sagebrush-grassland mosaic, southwestern Montana, USA. *Forest Ecology and Management* **230**, 107-118.

Heyerdahl EK, Brubaker LB, Agee JK (2001) Spatial controls of historical fire regimes: a multiscale example from the Interior West, USA. *Ecology* **82,** 660-678.

Holmes RL, Adams RK, Fritts HC (1986) Tree Ring Chronologies of western North America: California, eastern Oregon and northern Great Basin. Laboratory of Tree Ring Research Chronology Series VI (University of Arizona, Tucson, AZ)

Houston DB (1973) Wildfires in northern Yellowstone National Park. *Ecology* **54**, 1111-1117.

Huff MH (1995) Forest age structure and development following wildfires in the western Olympic Mountains, Washington. *Ecological Applications* **5**, 471-483.

Huisinga KD, Laughlin DC, Fulé PZ, Springer JD, McGlone CM (2005) Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. *Journal of the Torrey Botanical Society* **132**, 590-601.

Humphrey LD (1984) Patterns and mechanisms of plant succession after fire on *Artemisia*-grass sites in southeastern Idaho. *Vegetatio* **57**, 91-101

Isaac L (1940) Vegetation succession following logging in the Douglas-fir region with special reference to fire. *Journal of Forestry* **38**, 716-721.

Jameson DA (1962) Effects of burning on a galleta-black grama range invaded by juniper. *Ecology* **43**, 760-763.

Johnson DD, Miller, RF (2006) Structure and development of expanding western juniper woodlands as influenced by two topographic variables. *Forest Ecology and Management* **229**, 7-15.

Johnson EA, Miyanishi K, Bridge SRJ (2001) Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. *Conservation Biology* **15,** 1554-1557.

Keane RE, Veblen T, Ryan KC, Logan J, Allen C, Hawkes B (2002) The cascading effects of fire exclusion in the Rocky Mountains. Pages 133-153 *in* J. B. (Editor), editor. (Rocky Mountain Futures: An Ecological Perspective) (Island Press, Washington DC, USA)

Keeley JE (2002) Fire management of California shrubland landscapes. *Environmental Management* **29**, 395-408.
Keeley JE (2005) Chaparral fuel modification: what do we know --- and need to know? *Fire Management Today* **65(4),** 11-12.
Keeley JE, Fotheringham CJ (2003) Impact of past, present, and future fire regimes on North American Mediterranean shrublands, pp. 218-262. In T.T. Veblen, W.L. Baker, G. Montenegro, and T.W. Swetnam (eds), Fire and Climatic Change in Temperate Ecosystems of the Western Americas (Springer, New York)
Keeley JE, Fotheringham CJ, Baer-Keeley M (2005) Determinants of postfire recovery and succession in mediterranean-climate shrublands of California. Ecological Application*s* 15:1515-1534.
Keeley JE, Fotheringham CJ (2006) Wildfire management on a human dominated landscape: California chaparral wildfires, pp. 69-75. In G. Wuerthner, Editor**,** Wildfire ---A Century of Failed Forest Policy (Island Press, Covelo, CA)
Key CH, Benson NC (2005) Landscape Assessment: Ground measure of severity, the Composite Burn Index; and Remote sensing of severity, the Normalized Burn Ratio. *In* D.C. Lutes; R.E. Keane; J.F. Caratti; C.H. Key; N.C. Benson; S. Sutherland; and L.J. Gangi. (2005) *FIREMON: Fire Effects Monitoring and Inventory System*. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. Gen. Tech. Rep. RMRS-GTR-164-CD: LA1-51 (Fort Collins CO, USA)
Kitzberger T, Brown PM, Heyerdahl EK, Swetnam TW, Veblen TT (2007) Contingent Pacific-Atlantic Ocean influence on multicentury wildfire synchrony over western North America. *PNAS* **104**, 543-548.
Knapp PA, Soulé PT (1996) Vegetation change and the role of atmospheric CO_2 enrichment on a relict site in central Oregon: 1960-1994. *Annals of the Association of American Geographers* **86**, 387-411.
Knapp PA, Soulé PT, Grissino-Mayer HD (2001) Detecting potential regional effects of increased atmospheric CO2 on growth rates of western juniper. *Global Change Biology* **7**, 903-917.
Koch E (1942) History of the 1910 forest fires -- Idaho and western Montana. In: When the Mountains Roared: Stories of the 1910 fire. USDA Forest Service, Idaho Panhandle National Forest General Publication.
Kolb PF, Adams DL, McDonald GI (1998) Impacts of fire exclusion on forest dynamics and processes in central Idaho. *Tall Timbers Fire Ecology Conference* **20**, 911-923.
Koniak S (1985) Succession in pinyon-juniper woodlands following wildfire in the Great Basin. *Great Basin Naturalist* **45**, 556-566.
Laverty L, Williams J (2000) 'Protecting people and sustaining resources in fire-adapted ecosystems -- A cohesive strategy.' USDA Forest Service (Washington DC)
Lorimer CG, Gough WR (1988) Frequency of drought and severe fire weather in northeastern Wisconsin. *Journal of Environmental Management* **26**, 203-219.
Los Angeles Times, September 27, 1889. Los Angeles, California.
Malamud BD (2005) Characterizing wildfire regimes in the United States. *PNAS* **102**, 4694-4699.

Martin RL, Sapsis D (1991) Fires as agents of biodiversity: Pyrodiversity promotes biodiversity. In "Proceedings of the symposium on biodiversity of northwestern California" pp 150-157 (Wildland Resource Center, University of California, Berkeley)
McArthur ED (1979) Sagebrush systematics and evolution. Pages 14-22, *In* The sagebrush ecosystem: a symposium. Utah State University (Logan, UT, USA)
McArthur ED, Stevens R (2004) Composite shrubs. Pages 493-537, *In* Monsen, S.B.; Stevens, R.; Shaw, N.L., comps. Restoring western ranges and wildlands. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Gen. Tech. Rep. RMRS-GTR-136-vol-2 (Fort Collins, CO)
McDonough WT, Harniss RO (1974) Seed dormancy in *Artemisia tridentata* Nutt. *vaseyana* Rydb. *Northwest Science* **48**, 17-20.
McGarigal K, Marks BJ (1995) FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. USDA Forest Service Pacific Northwest Research Station General Technical Report PNW-GTR-351 (Portland, OR USA)
McNabb DH, Swanson FJ (1990) Effects of fire on soil erosion. In 'Natural and Prescribed Fire in Pacific Northwest Forests'. (Eds JD Walstad, SR Radosevich and DV Sandberg) pp. 159-176. (Oregon State University Press: Corvallis, OR)
Mensing SA, Michaelsen J, Byrne R (1999) A 560-year record of Santa Ana fires reconstructed from charcoal deposited in the Santa Barbara Basin, California. *Quaternary Research* **51**, 295-305.
Meyer SE, Monsen SB (1992) Big sagebrush germination patterns: Subspecies and population differences. *Journal of Range Management* **45**, 87-93.
Miller RF, Bates JD, Svejcar TJ, Pierson FB, Eddleman LE (2005) 'Biology, ecology, and management of western juniper' Oregon State University Agricultural Experiment Station Technical Bulletin 152 (Corvallis OR USA)
Miller RF, Heyerdahl E (2003) Fire regimes, pre- and post-settlement vegetation, and the modern expansion of western juniper at Lava Beds National Monument, California. US Department of Interior, National Park Service, Lava Beds National Monument, Final Report, CA.
Miller RF, Rose JA (1995) The historic expansion of western juniper in southeastern Oregon. *Great Basin Naturalist* **55**, 37-45.
Miller RF, Rose JA (1999) Fire history and western juniper encroachment in sagebrush steppe. *Journal of Range Management* **52,** 550-559.
Miller RF, Svejcar T, Rose, JA (1999) 'Conversion of shrub steppe to juniper woodland' Pages 385-390, *In* Monsen, S.B.; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Proc. RMRS-P-9 (Ogden, UT USA)
Miller RF, Svejcar T, Rose JA (2000) Impacts of western juniper on plant community composition and structure. *Journal of Range Management* **53**, 574-585.
Miller RF, Tausch RJ (2001) The role of fire in pinyon and juniper woodlands: a descriptive analysis. Tall Timbers. Pages 15-30, *In* K. Galley and T. Wilson (Eds.), Fire Conference 2000: The First National Congress On Fire, Ecology, Prevention And Management. Invasive Species Workshop: The Role of Fire In

The Control And Spread Of Invasive Species. Tall Timbers Research Station (Tallahassee, FL USA)
Minnich RA, Dezzani RJ (1991) Suppression, fire behavior, and fire magnitudes in Californian chaparral at the urban/wildland interface. Pages 67-83 in J. J. DeVries, editor. California watersheds at the urban interface, proceedings of the third biennial watershed conference. (University of California, Davis CA USA)
Mitchell JA, Robson D (1950) Forest fires and forest fire control in Michigan. Michigan Department of Conservation and USDA Forest Service (St. Paul, Minnesota) 82 pp.
Moreno, J.M. (Editor) (1998) Large Forest Fires. Backhuys Publishers, Leiden, The Netherlands. 498 pp.
Moritz MA, Keeley JE, Johnson EA, Schaffner AA (2004) Testing a basic assumption of shrubland fire management: Does the hazard of burning increase with the age of fuels? *Frontiers in Ecology and the Enviroment* **2**, 67-72.
Morrison PH, Swanson FJ (1980) Fire history and pattern in a Cascade Range landscape. USDA Forest Service Pacific Northwest Research Station General Technical Report PNW-GTR-254 (Portland, OR)
Mutch RW, Arno SF, Brown JK, Carlson CE, Ottmar RD, Peterson JL (1993) 'Forest health in the Blue Mountains: A management strategy for fire-adapted ecosystems' USDA Forest Service Pacific Northwest Research Station General Technical Report PNW-GTR-310 (Portland, OR USA)
Pernin RP (1999) The great Peshtigo fire: an eyewitness account, 2nd edition. State Historical Society of Wisconsin (Madison, Wisconsin, USA) 64 pp.
Peters EF, Bunting SC (1994) Fire conditions pre- and post-occurrence of annual grasses on the Snake River Plain. Pages 31-36, *In* Monsen, S.B.; Kitchen, S.G., comps. Proceedings – ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. U.S. Department of Agriculture, Forest Service, Intermountain Research Station Gen Tech. Rep. INT-GTR-313 (Ogden, UT)
Philpot CW (1974) The changing role of fire on chaparral lands. Pages 131-150 in M. Rosenthal, editor. Symposium on living with the chaparral, proceedings. Sierra Club (San Francisco, CA)
Philpot CW (1990) The wildfires in the northern Rocky Mountains and greater Yellowstone Area-1988. In 'Transactions of the Fifth-fifth North American Wildlife and Natural Resources Conference'. Denver, CO, USA. (Ed. RE McCabe) pp. 185-187
Pinol JK, Beven J, Viegas DX (2005) Modelling the effect of fire-exclusion and prescribed fire on wildfire size in Mediterranean ecosystems. *Ecological Modelling* **183**, 397-409.
Pollet J, Omi PN (2002) Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *International Journal of Wildland Fire* **11**, 1-10.
Pratt, SD, Holsinger L, Keane RE (2006) 'Modeling historical reference conditions for vegetation and fire regimes using simulation modeling' USDA Forest Service Rocky Mountain Research Station General Technical Report RMRS-GTR-175 (Fort Collins, CO USA)
Pregitzer KS, Saunders SC (1999) Jack pine barrens of the northern Great Lakes region. Pages 343-361 *in* Anderson, R. C., J. S. Fralish, and J. M. Baskin, editors.

Savannas, barrens, and rock outcrop plant communities of North America. (Cambridge University Press, Cambridge, U.K.)

Pyne SJ (1997) Fire in America: A cultural history of wildland and rural fire (University of Washington Press, Seattle, WA) 654 pp.

Radeloff VC, Mladenoff DJ, He HS, Boyce MS (1999) Forest landscape change in the northwest Wisconsin Pine Barrens from pre-European settlement to the present. *Canadian Journal of Forest Research* **29**, 1649-1659.

Radeloff VC, Mladenoff DJ, Boyce MS (2000) A historical perspective and future outlook on landscape scale restoration in the northwest Wisconsin pine barrens. *Restoration Ecology* **8**, 119-126.

Raymond CL, Peterson DL (2005) Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. *Canadian Journal of Forest Research* **35**, 2981-2995.

Reinkensmyer DP, Miller RF, Anthony RG (2007) Changes in avian communities along a mountain big sagebrush successional gradient. *J. of Wildlife Management* (in press).

Ricotta C, Avena G, Marchetti M (1999) The flaming sandpile: self-organized criticality and wildfires. *Ecological Modelling* **119**, 73-77.

Romme WH, Everham EH, Frelich LE, Moritz MA, Sparks RE (1998) Are large, infrequent disturbances qualitatively different from small, frequent disturbances? *Ecosystems* **1**, 524-534.

Romme WH, Floyd-Hanna L, Hanna D (2003) Ancient pinyon-juniper forests of Mesa Verde and the West: a cautionary note for forest restoration programs. Fire, fuel treatments, and ecological restoration: conference proceedings. USDA Forest Service Rocky Mountain Research Station Proceedings RMRS-P-O-29 (Ft. Collins CO USA)

Romme WH, Turner MG, Wallace LL, Walker JS (1995) Aspen, elk, and fire in Northern Yellowstone National Park. *Ecology* **76**, 2097-2106.

Ryan KC (2002) Dynamic interactions between forest structure and fire behavior in boreal ecosystems. *Silva Fennica* **36**, 13-39.

Savage M, Mast JN (2005) How resilient are southwestern ponderosa pine forests after crown fire? *Canadian Journal of Forest Research* **35(4)**, 967-977.

Schmidt WC, Shearer RC, Roe AL (1976) 'Ecology and silviculture of western larch forests.' USDA Forest Service, Technical Bulletin No. 1520, Washington DC.

Schmoldt DL, Peterson DL, Keane RE, Lenihan JM, McKenzie D, Weise D, Sandberg DV (1999) Assessing the effects of fire disturbance on ecosystems: a scientific agenda for research and management. USDA Forest Service Pacific Northwest Research Station General Technical Report GTR-PNW-455 (Portland OR)

Schoenberg FP, Peng R, Woods J (2003) On the distribution of wildfire sizes. *Environmetrics* **14**, 583-597.

Schoenberg, FP, Peng P, Huang Z, Rundel P (2003) Detection of non-linearities in the dependence of burn area on fuel age and climatic variables. *International Journal of Wildland Fire* **12(1),** 1-6.

Schoennagel TL, Veblen TT, Romme WH (2007) The interaction of fire, fuels, and climate across Rocky Mountain landscapes. *Bioscience* **54**, 651-672.

Schulte LA, Mladenoff DJ (2005) Severe wind and fire regimes in northern Wisconsin (USA) forests: historical variability at the regional scale. *Ecology* **86**, 431-445.

Schulte LA. Mladenoff DJ, Burrows SN, Sickley TA, Nordheim EV (2005) Spatial controls of Pre-Euroamerican wind and fire in Wisconsin (USA) forests: a multiscale assessment. *Ecosystems* **8**, 73-94

Schulte LA, Mladenoff DJ, Crow TR, Merrick L, Cleland DT (2007) Homogenization of northern U.S. Great Lakes forests as a result of land use. *Landscape Ecology.*

Simard AJ, Blank RW (1982) Fire history of a Michigan jack pine forest. *Michigan Academician* **15**, 59-71.

Soulé PT, Knapp PA, Grissino-Mayer HD (2004) Human agency, environmental drivers, and western juniper establishment during the late Holocene. *Ecological Applications* **14**, 96-112

Sturtevant BR, Zollner PA, Gustafson EJ, Cleland DT (2004) Human influence on the abundance and connectivity of high-risk fuels in mixed forests of northern Wisconsin, USA. *Landscape Ecology* **19**, 235-253.

Spies TA, Franklin JF, Thomas TB (1988) Coarse woody debris in Douglas-fir forests of western Oregon and Washington. *Ecology* **69**, 1689-1702.

Spies TA, Ripple WJ, Bradshaw GA (1994) Dynamics and pattern of a managed coniferous forest landscape in Oregon. *Ecological Applications* **4**, 555-568.

State of Oregon (1997) Tillamook burn to Tillamook State Forest. Oregon Department of Forestry (Salem, OR)

Stephens SL (2005) Forest fire causes and extent on United States Forest Service lands. *International Journal of Wildland Fire* **14,** 213-222.

Strauss D, Bednar L, Mees R (1989) Do one percent of forest fires cause ninety-nine percent of the damage? *Forest Science* **35**, 319-328.

Strom BA (2005) Pre-fire treatment effects and post-fire forest dynamics on the Rodeo-Chediski burn area, Arizona. M.S. Thesis, School of Forestry, Northern Arizona University, (Flagstaff AZ USA)

Strom BA, Fulé PZ. In press. Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. *International Journal of Wildland Fire*.

Swetnam TW, Baisan CH (2003) Tree-ring reconstructions of fire and climate history in the Sierra Nevada and southwestern United States. Pages 158-195 *in* T. T. Veblen, W. L. Baker, G. Montenegro, and T. W. Swetnam, editors. Fire and climatic change in temperate ecosystems of the western Americas (Springer-Verlag, New York, New York, USA)

Swetnam TW, Betancourt JL (1997 Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. *Journal of Climate* **11**, 3128-3147.

Tappeiner JC, Huffman D, Marshall D, Spies TA, Bailey JD (1997) Density, tree ages, and growth rates in old-growth and young-growth forests in coastal Oregon. *Canadian Journal of Forest Research* 27, 638-648.

Tausch RJ (1999) Historic pinyon and juniper woodland development. Pages 12-19, *In* Monsen, S.B.; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18;

Provo, UT. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Proc. RMRS-P-9 (Ogden, UT)

Tausch RJ, Nowak CL (2000) Influences of Holocene climate and vegetation changes on present and future community dynamics. *Journal of Arid Land Studies* **10S**, 5-8.

Tausch RJ, West NE, Nabi AA (1981) Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. *Journal of Range Management* **46**, 439-447.

Tausch RJ (1999a) Historic woodland development. Pgs. 12-19. *in* S.B. Monsen, R. Stevens, R.J. Tausch,, R. Miller and S. Goodrich (eds.). Proceedings: ecology and management of pinyon-juniper communities within the Interior West. USDA Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-9 (Ogden, UT)

Tausch RJ (1999b) Transitions and threshold: influences and implications for management in pinyon and Utah juniper woodlands. Pg. 61-65. *in* S.B. Monsen, R. Stevens, R.J. Tausch,, R. Miller and S. Goodrich (eds.). Proceedings: ecology and management of pinyon-juniper communities within the Interior West. USDA Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-9 (Ogden, UT)

Taylor AH, Skinner CN (1998) Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. *Forest Ecology and Management* **111**, 285-301.

Taylor AH, Skinner CN (2003) Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. *Ecological Applications* **13**, 704-719.

Teensma PDA (1987) Fire history and fire regimes of the central western Cascades of Oregon. Ph.D. dissertation (University of Oregon, Eugene, OR)

Turner MG, Baker WL, Peterson CJ, Peet RK (1998) Factors influencing succession: lessons from large, infrequent natural disturbances. *Ecosystems* **1**, 511-523.

Turner MG, Romme WH, Gardner RH (1999) Prefire heterogenity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. *International Journal of Wildland Fire* **9**, 21-36.

Turner MG, Romme WH, Tinker DB (2003) Surprises and lessons from the 1988 Yellowstone fires. *Front Ecol Environ* **1**, 351-358.

USDA, Forest Service, Humboldt Toyiabe National Forest, Bridgeport Ranger District. 1999. Internal report.

van Wagtendonk JW (1995) Large fires in wilderness areas. In 'Proceedings : Symposium on Fire in Wilderness and Park Management : Missoula, MT, March 30-April 1, 1993.' pp. 113-116. (Ogden UT : U.S. Dept. of Agriculture Forest Service Intermountain Research Station 1995)

Wadleigh L, Jenkins MJ (1996) Fire frequency and the vegetative mosaic of a spruce-fir forest in northern Utah. *Great Basin Naturalist* **56**, 28-37

Waichler WS, Miller RF, Doescher PS (2001) Community characteristics of old-growth western juniper woodlands in the pumice zone of central Oregon. *Journal of Range Management* **54**, 518-527.

Wambolt CL, Hoffman TL, Mehus CA (1999) Response of shrubs in big sagebrush habitats to fire on the Yellowstone winter range. Pages 238-242, *In* McArthur, E.D.; Ostler, W.K.; Wambolt, C.L., comps. Proceedings: shrubland ecotones;

1998 August 12-14; Ephraim, UT. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Proc. RMRS-P-11 (Ogden, UT)
Wambolt CL, Walhof KS, Frisina MR (2001) Recovery of big sagebrush communities after burning in south-western Montana. *Journal of Environmental Management* **61,** 243-252.
Wangler MJ, Minnich RA (1996) Fire and succession in pinyon-juniper woodlands of the San Bernardino Mountains, California. *Madroño* **43**, 493-514.
Watts MJ, Wambolt CL (1996) Long-term recovery of Wyoming big sagebrush after four treatments. *Journal of Environmental Management* **46**, 95-102.
Weisberg PJ (2004) Importance of non-stand-replacing fire for development of forest structure in the Pacific Northwest, USA. *Forest Science* **50**, 245-258.
Weisberg P, Swanson F (2003) Regional synchroneity of fire regimes of the western Cascades, USA. *Forest Ecology and Management* **172**, 17-28.
Welch BL (2005) Big sagebrush: A sea fragmented into lakes, ponds, and puddles' U.S. Department of Agriculture Forest Service Rocky Mountain Research Station Gen. Tech. Rep. RMRS-GTR-144 (Fort Collins, CO) 210 p.
Wellner CA (1970) Fire history in the northern Rocky Mountains. In 'Role of Fire in the Intermountain West'. Missoula, MT pp. 42-64
West NE (1999) Juniper-pinyon savannas and woodland of western North America. Pages 288-308 *in* R.C. Anderson, J.S Fralish, and J.M. Baskin (eds.) Savannas, barrens, and rock outcrop plant communities of North America (Cambridge University Press, London, United Kingdom)
West NE, Yorks TP (2002) Vegetation responses following wildfire on grazed and ungrazed sagebrush semi-desert. *Journal of Range Management* **55**, 171-181.
West NE, Van Pelt NS (1986) Successional patterns in pinyon-juniper woodlands. Pages 43-52 *in* R.L. Everett (compiler). Proceedings: pinyon-juniper conference. USDA Forest Service, Intermountain forest and Range Experiment Station, General Technical Report INT-215 (Ogden, UT)
Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW (2006) Warming and earlier spring increase western U.S. forest wildfire activity. *Science* **313**, 940-943
White MA, Vankat JK (1993) Middle and high elevation coniferous forest communities of the North Rim region of Grand Canyon National Park, Arizona, USA. *Vegetatio* **109**, 161-174.
Whisenant SG (1990) Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. Pages 4-10, *In* McArthur, E.D.; Romney, E.M.; Smith, S.D.; Tueller, P.T., comps. Proceedings – symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management: 1989 April 5-7; Las Vegas, NV U.S. Department of Agriculture, Forest Service, Intermountain Research Station Gen Tech. Rep. INNT-276 (Ogden, UT)
Whitney GG (1986) The relation of Michigan's presettlement pine forests to substrate and disturbance history. *Ecology* **67**, 1548-1559.
Will-Wolf S, Stearns F (1999) Dry soil oak savanna in the Great Lakes region. Pages 135-153 *in* Anderson, R. C., J. S. Fralish, and J. M. Baskin, editors. Savannas, barrens, and rock outcrop plant communities of North America (Cambridge University Press, Cambridge, U.K.)

Wimberly MC, Spies TA, Long CJ, Whitlock C (2000). Simulating historical variability in the amount of old forests in the Oregon Coast Range. *Conservation Biology* **14**, 167-180.
Woodbury AM (1947) Distribution of pigmy conifers in Utah and northeastern Arizona. *Ecology* **28**:113-126.
Wright HA, Neuenschwander LF, Britton CM (1979) The role and use of fire in sagebrush-grass and pinyon-juniper plant communities. a state-of-the-art review. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station Gen. Tech. Rep. INT-58 (Ogden, UT) 48 p.
Wright HA, Bailey AW (1982) Fire ecology, United States and southern Canada (John Wiley & Sons, New York) 501 p.
Young JA, Evans RA (1981) Demography and fire history of a western juniper stand. *Journal of Range Management* **34**, 501-505.
Young JA, Evans RA (1989) Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. *Weed Science* **37**, 201-206.
Zhang Q, Pregitzer KS, Reed DD (1999) Catastrophic disturbance in the presettlement forest of the Upper Peninsula of Michigan. *Canadian Journal of Forest Research* **29**, 106-114.

Tables

Table 1. Examples of fire return intervals for Pacific Northwest forests. Sites are listed from north to south in the region.

Forest Type/Location	Fire return Interval (yrs)	Source
Western Washington	230	Fahnestock and Agee (1983)
Mount Rainier, WA	434	Hemstrom and Franklin (1982)
Bull Run, OR	350	Agee and Krusemark (2001)
Central Oregon Cascades	95-145	Morrison and Swanson (1990)
Central Oregon Cascades	100	Teensma (1987)
Siskiyou Mountains	15-75	Agee (1991)
Siskiyou Mountains	12-19	Taylor and Skinner (1998)
Klamath Mountains	11.5-16.5	Taylor and Skinner (2003)

Table 2 – A comparison of burn severity and patch metrics (mean and standard deviation in parenthesis) across fires less than 10,000 ha (small fires) and fires > 10,000 ha (large fires). P-values in bold indicate significance at 0.05 level using t-test statistics.

Attribute	***Small Fires***	***Large Fires***	***P value***
Number of Fires	25	11	--
Burn Severity			
Unburned	0.21 (0.0139)	0.15 (0.0047)	0.10465
Low	0.25 (0.0086)	0.21 (0.0052)	0.27295
ModerateLow	0.18 (0.0091)	0.18 (0.0013)	0.27295
Moderate-High	0.19 (0.0091)	0.20 (0.0066)	0.70221
High	0.16 (0.0162)	0.25 (0.0113)	0.05251
Patch Metrics			
Patch density (patches 100 ha^{-1})	91.2 (24.5)	66.8 (5.78)	**0.00272**
Largest patch index (dimensionless)	16.9 (9.25)	10.7 (7.48)	0.05812
Landscape shape index (dimensionless)	23.1 (9.59)	103.2 (45.44)	**0.00001**
Shape index (dimensionless)	5.08 (2.97)	8.85 (4.13)	**0.00387**
Fractal index (dimensionless)	1.06 (0.01)	1.06 (0.01)	0.35577
Edge density (m ha-1)	306.4 (49.8)	247.5 (13.3)	**0.00052**
Contagion (0-100 index)	34.0 (5.05)	35.4 (2.07)	0.38853
Interspersion-juxtaposition index (0-100)	64.1 (5.90)	61.8 (1.99)	0.18497

Table 3 – A summary of large fire characteristics as described in this paper (Y-yes, N-No, M-more, L-less, S-same)

Ecosystem	*Historically common?*	*Currently more severe?*	*Currently more frequent?*	*Important to ecosystem?*	*Pre-conditioning factors*
Pacific Northwest	**Y**	**S**	**S**	**Y**	**Drought**
Southern California	**Y**	**S**	**S**	**Y**	**Santa ana winds**
Northern Rockies	**Y**	**S**	**S**	**Y**	**Drought, wind**
Southwestern US	**Y**	**M**	**S**	**Y**	**Drought**
Sagebrush-Grasslands	**N**	**M**	**M**	**N**	**Fuel contagion**
Pinon-Juniper	**Y**	**S**	**S**	**N**	**Wind, Hot Weather**
Great Lakes Mixed forests	**Y**	**N**	**N**	**Y**	**Drought, Ignition**

Figures

Figure Captions

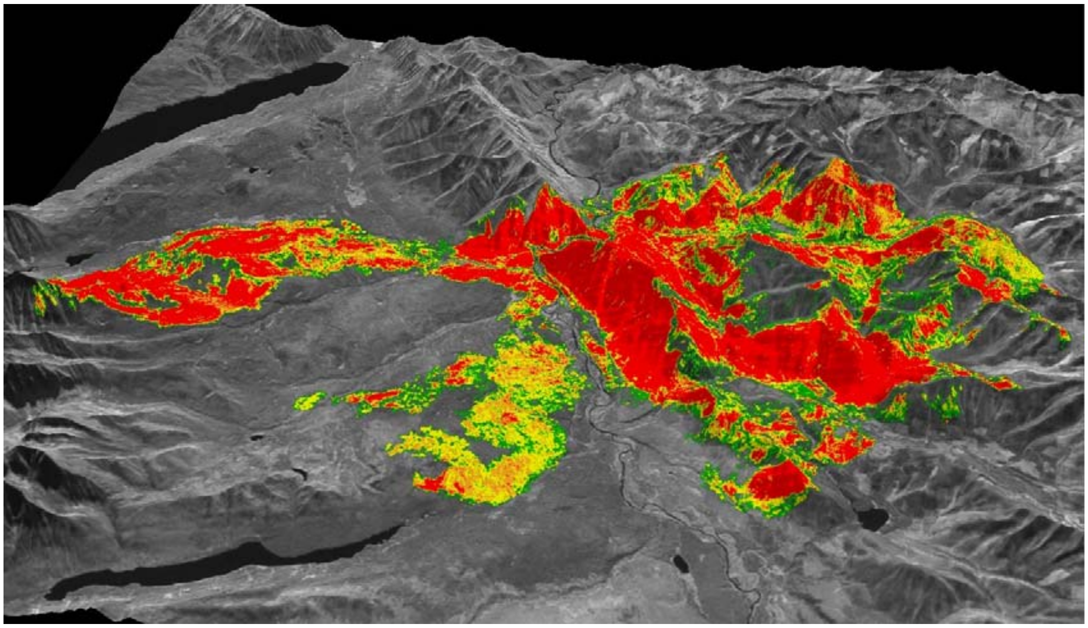
Figure 1. A map of burn severity for the Moose fire near Glacier National Park, Montana, USA showing the complex mosaic left behind by a large fire in the northern Rocky Mountains. Highest severities are in red while lowest are in greens and yellows.

Figure 2. The fire history of the area within the Cedar and Otay fires showing the great diversity of stand ages and sizes.

Figure 3. A large stand replacement fire in sagebrush grasslands facilitated by abundant fine fuels due to cheatgrass invasions (photo by Stan Kitchen).

Figure 4. A landscape after a large stand replacement fire in a closed Utah juniper and two-needle piñon pine woodland on the Kiabab Plateau in Arizona (photo by Rick Miller).

Figure 1. A map of burn severity for the Moose fire near Glacier National Park, Montana, USA. Highest intensities are in red while lowest are in greens and yellows.



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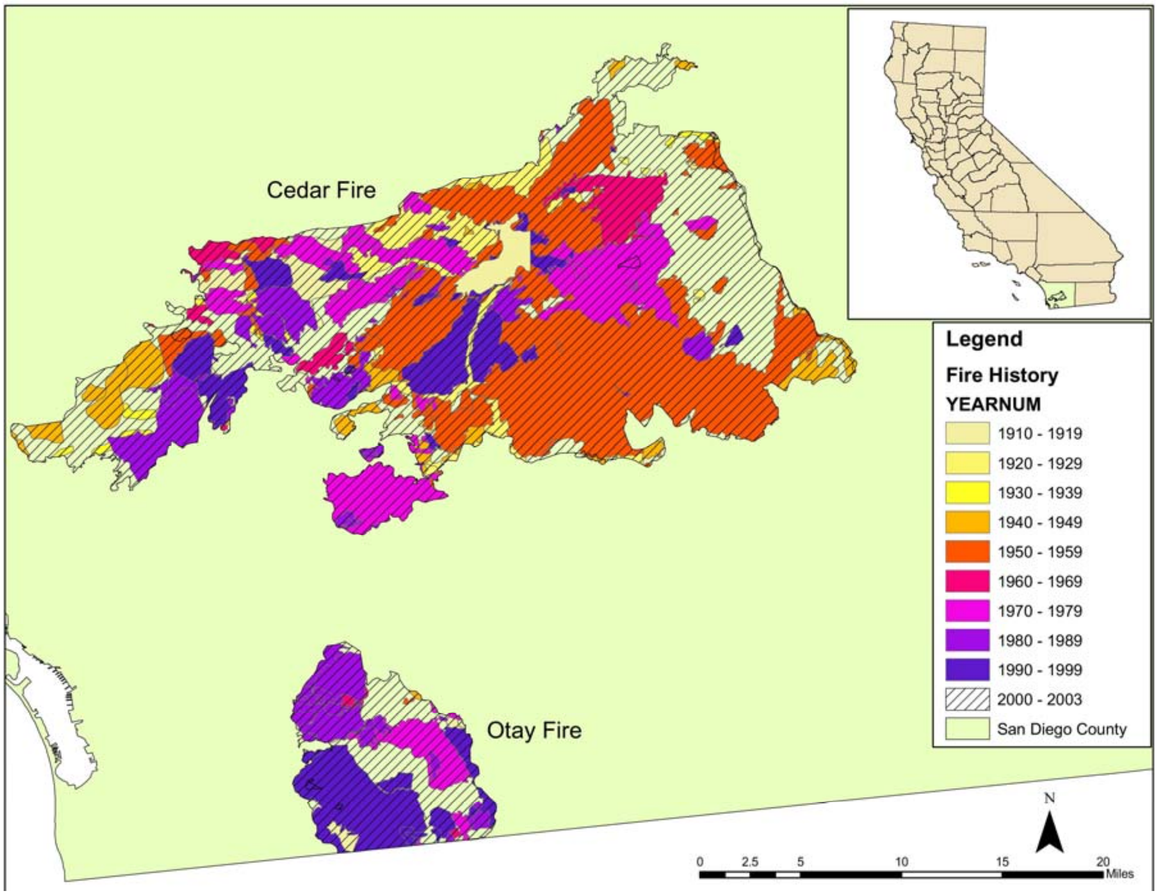

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